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RISK ASSESSMENT WORK PLAN ADDENDUM
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FERNALD ENVIRONMENTAL MANAGEMENT PROJECT
FERNALD, OHIO

REMEDIAL INVESTIGATION and FEASIBILITY STUDY

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U.S. DEPARTMENT OF ENERGY
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LIST OF ACRONYMS

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AEC	U.S. Atomic Energy Commission
ARAR	applicable or relevant and appropriate requirement
BEIR	Biological Effects of Ionizing Radiation
CEDE	committed effective dose equivalent
CERCLA	Comprehensive Environmental Response Compensation and Liability Act
CFR	Code of Federal Regulations
CIS	Characterization and Investigation Study
CLP	Contract Laboratory Program
CV	coefficient of variation
DOE	U.S. Department of Energy
EP	extraction procedure
EPA	U.S. Environmental Protection Agency
FEMP	Fernald Environmental Management Project
FFCA	Federal Facility Compliance Agreement
FMPC	Feed Materials Production Center
HEAST	Health Effects Assessment Summary Tables
HI	Hazard Index
HQ	Hazard Quotient
IRIS	Integrated Risk Information System
ISA	Initial Screening of Alternatives
LOEC	lowest observable effects concentrations
MDL	minimum detection limit
MGD	million gallons per day
MSL	mean sea level
MT	metric tons
MUSLE	Modified Universal Soil Loss Equation
ND	nondetection
NESHAPS	National Emission Standards for Hazardous Air Pollutants
NLO	National Lead Company of Ohio
NOEC	no observable effects concentration
NPDES	National Pollutant Discharge Elimination System
NRC	U.S. Nuclear Regulatory Commission
OEPA	Ohio Environmental Protection Agency
OSHA	Occupational Safety and Health Administration
PCB	polychlorinated biphenyl

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QA/QC	quality assurance/quality control
QAPP	Quality Assurance Project Plan
RA	Risk Assessment
RCRA	Resource Conservation and Recovery Act
RfD	reference dose
RI/FS	remedial investigation/feasibility study
RME	reasonable maximum exposure
RMI	Reactive Metals Incorporated
ROD	Record of Decision
SARA	Superfund Amendments and Reauthorization Act
SF	Shielding Factor
TCLP	Toxicity Characteristic Leaching Procedure
TIC	tentatively identified compounds
USLE	Universal Soil Loss Equation
UTL	upper tolerance limit
VOC	volatile organic compound
WEMCO	Westinghouse Environmental Management Company of Ohio
WMCO	Westinghouse Materials Company of Ohio

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APPLICABLE OR RELEVANT AND APPROPRIATE REQUIREMENTS (ARARs) - Requirements set forth in regulations that implement environmental and public health laws and must be attained or exceeded by a selected remedy, unless a waiver is invoked. ARARs are divided into three categories: chemical-specific, location-specific, and action-specific, depending on whether the requirement is triggered by the presence or emission of a chemical, by a vulnerable or protected location, or by a particular action.

AQUIFER - An underground geological formation, group of formations, or part of a formation that is capable of yielding a significant amount of water to a well or spring.

BASELINE RISK ASSESSMENT - The studies undertaken for Operable Units (OUs) 1-5 to characterize the current and potential threats to human health and the environment that may be posed by contaminants within those operable units. Each Baseline Risk Assessment shall provide a framework for developing risk information necessary to assist in developing remedial alternatives, and shall consider the risks that currently exist at the site, if no further response actions or institutional controls are applied. There are four steps in the baseline risk assessment process: data collection and analysis; exposure assessment; toxicity assessment; and risk characterization. The baseline risk assessment contributes to the site characterization and subsequent development, evaluation, and selection of appropriate response alternatives.

CHRONIC REFERENCE DOSE (RfD) - An estimate (with uncertainty spanning perhaps an order of magnitude or greater) of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime. Chronic RfDs are specifically developed to be protective for long-term exposure to a compound (as a Superfund program guideline, seven years to lifetime).

COMPREHENSIVE RESPONSE ACTION RISK EVALUATION- An evaluation that shall be developed for each OU and included as an appendix to the applicable FS Reports. Each Comprehensive Response Action Risk Assessment will evaluate the risk associated with the proposed alternatives and factor in the cumulative residual risk associated with the other OUs. The purpose of this analysis is to evaluate the potential risk reduction from each proposed alternative in the context of the risk posed by the site as a whole. The cumulative residual risk contribution from the other OUs will be estimated based upon the selected alternative, or the Leading Remedial Alternative, which will be initially presented in the Site-Wide Characterization Report.

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COMPREHENSIVE SITE-WIDE OPERABLE UNIT - An evaluation of remedies selected for OUs 1-5, including remedial and removal actions, to ensure that they are protective of human health and the environment on a site-wide basis, as required by CERCLA, the NCP and applicable U.S. EPA policy and guidance. The Comprehensive Site-Wide Operable Unit shall include a Remedial Investigation/Projected Residual Risk Assessment Report, a Proposed Plan and Record of Decision (ROD) which provide that no additional action is necessary to achieve protectiveness, or if necessary, a Site-Wide Feasibility Study, Proposed Plan and ROD.

CONCEPTUAL MODELS - Models that are constructed to describe or represent various phenomena under a specific set of conditions, or assumptions to estimate the resultant effect(s). As applied to risk assessment, conceptual models are used as a basis for calculational fate and transport analysis and exposure assessment. Standard industry accepted calculational model (computer-codes) are utilized for this purpose under FEMP RI/FS.

CONSENT AGREEMENT - An Agreement between the U.S. EPA and the U.S. DOE for the cleanup of the FEMP under authorities of Sections 106 and 120 of Superfund Amendments and Reauthorization Act of 1986 (SARA). The Consent Agreement signed in April 1990, amends the July 1986 Federal Facility Compliance Agreement (FFCA), which established the original framework for the FMPC environmental investigation and cleanup. A modified Consent Agreement, signed in September 1991, including renegotiated framework and schedules for developing, implementing, and monitoring appropriate response actions at the site and to facilitate cooperation, exchange of information and participation of the Parties in such actions.

CONTAMINANTS OF POTENTIAL CONCERN - Chemicals and radionuclides that are potentially site-related and whose data are of sufficient quality for use in the quantitative risk assessment.

CRITICAL SUBPOPULATION - Populations at high potential risk from radionuclide or chemical exposure due to increased sensitivity, special behavior patterns, and/or current or past exposures from other sources. Critical subpopulations include infants and children, the elderly, pregnant and nursing women, individuals with chronic illnesses, and individuals previously exposed to chemicals or radionuclides during occupational activities or by residing industrial areas.

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CURRENT LAND USE - One of the general categories of use of real property at a site that realistically describes the current use of the property for purposes of assessing potential human health risks. These categories include: residential, agricultural, commercial/industrial; and recreational.

ECOLOGICAL RISK ASSESSMENT - A site-specific analysis of the potential risks (current and future) to ecological receptors. The ecological risk assessment determines whether facility-derived constituents in environmental media on or adjacent to the facility, currently have or may potentially have adverse ecological impacts. Also referred to as an environmental risk assessment.

EXPOSURE ASSESSMENT - The determination or estimation (qualitative or quantitative) of the magnitude, frequency, duration, and route of exposure.

EXPOSURE PATHWAY - The course a chemical or physical agent takes from a source to a receptor organism. Each exposure pathway includes a source or release from a source, an exposure point, an exposure route, and a receptor. If the exposure point differs from the source, a transport medium (e.g., air) or media (in cases of intermedia transfer) also is included.

EXPOSURE SCENARIO - A chain of events and conditions defining a combination of exposure pathways and processes that are used to estimate reasonable maximum exposure of individuals or groups.

FATE AND TRANSPORT MODELING - Modeling used to assess contaminant movement from source areas to receptor locations through various media (e.g., groundwater, air). Used in conjunction with monitoring data, these models estimate contaminant concentrations at exposure point locations where measured contaminant concentration data is not available, such as off-property locations, or contaminant distribution in the future.

FEASIBILITY STUDY (FS) - The study that fully evaluates and develops remedial action alternatives to prevent or mitigate the migration or release of hazardous substances, pollutants, contaminants, or hazardous constituents at and from the site. The FS is generally performed in conjunction with the remedial investigation (RI) and uses data gathered during the RI to develop remedial action alternatives, and to undertake an initial screening and detailed analysis of the alternatives. The RI data are used to define the objectives of the response action, to develop

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remedial action alternatives, and to undertake an initial screening and detailed analysis of the alternatives. The FS includes a report that describes remedial action alternatives and documents the selection process.

FEMP - The Fernald Environmental Management Project, the present name for the former Feed Materials Production Center in Fernald, Ohio, starting August 23, 1991.

FMPC - The former Feed Materials Production Center in Fernald, Ohio, which is now renamed the Fernald Environmental Management Project on August 23, 1991 to reflect the change in its mission from that of a production facility to an environmental restoration project.

FUTURE POTENTIAL LAND USE - The hypothesized use of property at a site that describes plausible use of the property in the future for purposes of assessing potential human health risks. These categories may include: residential; agricultural; commercial/industrial; and recreational.

GROUNDWATER - Water in a saturated zone or stratum beneath the surface of land or water.

INSTITUTIONAL CONTROLS - Measures that generally limit human activities at or near facilities where hazardous substances, pollutants, or contaminants exist or will remain on site. Active institutional controls include engineering controls and an active security program. Passive institutional controls include monuments, land and resource restrictions, deed restrictions, permitting programs, zoning, government ownership, and deed notices. Institutional controls may supplement engineering controls (e.g., treatment and/or containment of source material) to provide protection of human health.

INTAKE - A measure of exposure. For chemicals, it is expressed as the mass of a chemical in contact with the exchange boundary of a receptor per unit body weight per unit time (e.g., mg chemical/kg body weight-day). For radionuclides, it is expressed as the activity of a radionuclide (e.g., Bq or Ci) taken into an organism. Intake by inhalation, ingestion and dermal absorption are the three most important exposure routes for both chemicals and radionuclides.

LEADING REMEDIAL ALTERNATIVE - The remedial alternative which, based upon all available data and best professional judgement, is the most likely to be selected as the response action for an OU. The Leading Remedial Alternative does not represent the pre-selection of a

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remedy and shall be used only for the purpose of estimating and evaluating the risk presented by the entire Site during the FS/Comprehensive Response Action Risk Assessments for OUs 1-5. The Leading Remedial Alternative shall be modified as necessary to reflect new data and information and shall in no way prescribe or restrict the selection of the remedy for the OU 1-5 RODs.

ON-SITE - The areal extent of contamination and all suitable areas in very close proximity to the contamination necessary for implementation of the response action.

OPERABLE UNIT - A discrete action that comprises an incremental step toward comprehensively addressing Site problems.

PERCHED GROUNDWATER - Groundwater within the glacial overburden that is present in isolated pockets or zones; that is distinct from the regional aquifer; and that contains a limited volume of water.

POINTS OF COMPLIANCE - All appropriate locations in the media of concern at a site where remediation goals are to be attained. The points of compliance also define the locations from which a sample or set of samples could be selected for the purpose of monitoring the progress of remediation activities or for determining when chemical-specific remediation goals have been achieved.

POINT OF DEPARTURE - The risk level of 10^{-6} that is used as the starting point (or initial "protectiveness" goal) for determining the most appropriate risk level that alternatives should be designed to attain as described in 40CFR300.430(e)(9)(iii).

REASONABLE MAXIMUM EXPOSURE (RME) - The exposure that is reasonably expected to occur at a site under both current and future land-use conditions and defined by conservative exposure parameters. The intent of the RME is to estimate a conservative exposure case (i.e., well above the average case) that is still within the range of possible exposures. It does not embrace all hypothetical possibilities, but rather is limited to situations and conditions that "are likely to occur". RMEs are estimated for individual pathways. If a population is potentially exposed via more than one pathway, an RME must be estimated for the combination of pathways.

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RECEPTOR - A member of human, animal, or plant populations that may be exposed to radioactive or hazardous materials.

REMEDIAL ACTION - A comprehensive response action that provides a permanent remedy to mitigate risks associated with hazardous waste under CERCLA and to remedy any condition that could lead to future risks. A remedial action should include a monitoring system to ensure that such action protects the public health and welfare and the environment and, where appropriate, to confirm post-removal site control activities.

REMEDIAL ACTION OBJECTIVES (RAOs) - Site-specific, quantitative goals that define the extent of cleanup required to achieve CERCLA response objectives. RAOs specify contaminants of concern, media of concern, potential exposure pathways, and remediation goals for the site.

REMEDIATION GOALS (RGs) - A subset of RAOs that specify the allowable concentration of each contaminant of concern in each environmental medium of concern that should be achieved by a remediation effort. Preliminary remediation goals are developed based on readily available information such as chemical-specific ARARS (e.g., MCLs) or other reliable information. Preliminary remediation goals are modified, as necessary, as more information becomes available during the RI/FS. Final remediation goals are determined when the remedy is selected.

REMEDIAL INVESTIGATION (RI) - The investigation conducted to fully determine the nature and extent of the release or threat of release of hazardous substances, pollutants, contaminants, or hazardous constituents. The RI emphasizes data collection and site characterization. The RI includes sampling and monitoring, as necessary, and includes the gathering of sufficient information to support the Feasibility Studies and the risk assessments.

REMOVAL ACTION - The cleanup or removal of released hazardous substances from the environment taken in the event of the imminent threat of release of hazardous substances into the environment.

RESPONSE ACTION - The action that encompasses all response measures, including removal action and remedial action, consistent with the National Contingency Plan, to reduce the imminent threat of release of hazardous substances into the environment (removal action) and/or

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to provide a permanent remedy to mitigate risks associated with hazardous substances and to remedy any condition that could lead to future risks (remedial action) to protect the public health or welfare or the environment.

RISK CHARACTERIZATION - The part of the risk assessment that summarizes and combines outputs of the exposure and toxicity assessments to characterize baseline risk, both in quantitative expressions and qualitative statements. During risk characterization, chemical-specific toxicity information is compared against both measured contaminant exposure levels and those levels predicted through fate and transport modeling to determine whether current or future risk levels at or near the site are of potential concern.

SEDIMENT - The unconsolidated inorganic and organic material that is suspended in and is transported by surface water, or has settled out and has deposited into beds.

SITE - Areas within the property boundary of FEMP and any other areas that received or potentially received released hazardous substances, pollutants, contaminants, or hazardous constituents. The term shall have the same meaning as "facility" as defined by Section 101(9) of CERCLA, 42 U.S.C. § 9601(9).

SITE-WIDE BASELINE RISK ASSESSMENT - The baseline risk assessment that includes contributions to potential adverse health effects (current or future) from the entire site (including all operable unites).

SITE-WIDE CHARACTERIZATION REPORT - A one time summary of all site data available as of December 1, 1991. Based upon this data, and upon best professional judgement, U.S. DOE shall present Leading Remedial Alternatives for OUs 1-5. Additionally, this report shall contain a Preliminary Baseline Risk Assessment which characterizes the current and potential threats to human health and the environment that may be posed by contaminants at the entire Site. The Preliminary Baseline Risk Assessment shall consider the risks which currently exist at the Site, if no further response actions or institutional controls are applied.

SITE-WIDE FEASIBILITY STUDY (SITE-WIDE FS) - A study undertaken in the event U.S. EPA determines that further remedial actions, are necessary to ensure protection of human health and the environment as documented in the Site-Wide RI/Projected Residual RA. This

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study shall fully evaluate and develop remedial action alternatives which, in conjunction with the remedial and removal actions previously taken or selected at the Site, ensure that response actions are protective of human health and the environment. However, if U.S. EPA determines that the results of the Site-Wide RI/Projected Residual RA Report indicate that the selected removal and remedial alternatives for OUs 1-5 are protective of human health and the environment on a site-wide basis, a Site-Wide FS Study will not be required.

SITE-WIDE REMEDIAL INVESTIGATION/PROJECTED RESIDUAL RISK ASSESSMENT REPORT (SITE-WIDE RI/PROJECTED RESIDUAL RA REPORT) - A report prepared following finalization of the RODs for OUs 1-5. The Site-Wide RI shall incorporate by reference all data collected pursuant to the RIs for OUs 1-5 or the removal actions and shall summarize any data collected after finalization of the OU 1-5 RODs. The Site-Wide RI shall also gather any additional sampling data if necessary to support the Site-Wide Feasibility Study. Additionally, the Projected Residual RA shall document all risk which is anticipated to remain at the Site following the implementation of the selected response actions embodied in the OU 1-5 and the selected removal actions. The Projected Residual Risk Assessment shall be used to determine whether the previously selected response actions are protective of human health and the environment as required by CERCLA, the NCP and applicable U.S. EPA policy and guidance.

SITE-WIDE RESIDUAL RISK ASSESSMENT - A site-specific analysis of the potential adverse health effects that could be caused by hazardous substances that remain at the Site (including all operable units) after completion of all response actions at the Site. The concentrations that are used to calculated the risks are the final actually measured concentrations of the contaminants that remain at the Site, which include "new" chemicals that were not previously identified during the baseline risk assessment, but that may have resulted from the remedial actions.

SLOPE FACTOR - A plausible upper-bound estimate of the probability of a response per unit intake of a chemical or radionuclide over a lifetime. The slope factor is used to estimate an upper-bound probability of an individual developing cancer as a result of a lifetime of exposure to a particular level of a potential carcinogen.

SOIL - All unconsolidated materials normally found on or near the surface of the earth including, but not limited to, silts, clays, sands, gravel, and small rocks.

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SURFACE WATER - All water that is open to the atmosphere and subject to surface runoff.

TOXICITY ASSESSMENT - The part of the baseline risk assessment that considers: 1) the types of adverse health effects associated with chemical exposures; 2) the relationship between magnitude of exposure and adverse effects; and 3) related uncertainties such as the weight of evidence of a particular chemical's carcinogenicity in humans.

WORK PLAN ADDENDUM - A supplement to the RI/FS Work Plan that established the scope and specific methodology for risk assessment and risk management activities in the RI and FS.

1.0

1.0 INTRODUCTION

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In accordance with the provisions of the Amended Consent Agreement, dated September 1991, between the U.S. Environmental Protection Agency (EPA) and the U.S. Department of Energy (DOE), a methodology has been prepared for performing risk assessments and establishing risk-based remedial action goals at the Fernald Environmental Management Project (FEMP) (formerly the Feed Materials Production Center [FMPC]). This addendum to the Remedial Investigation/Feasibility Study (RI/FS) Work Plan for the FEMP presents this methodology and has been prepared to fulfill the requirements of Section X, Paragraph B.1, of the Amended Consent Agreement.

1.1 OBJECTIVES OF WORK PLAN ADDENDUM

This Work Plan Addendum has been prepared to achieve the following three objectives: (1) establish specific risk assessment methodology to be followed in RI and FS risk assessment work for the FEMP; (2) establish the scope of risk assessment work; and (3) document the specific approach to be followed when determining whether estimated risks associated with selected remedial alternatives for the entire site are protective of human health and the environment.

The RI/FS work performed to date at the FEMP has revealed key technical issues and programmatic uncertainties that have hampered the document review and approval process. Efforts to resolve key technical issues hindering completion of the RI/FS process are ongoing. It is intended that this Work Plan Addendum address and effect resolution of those technical issues pertaining to risk assessment. One of the goals of this addendum is to secure EPA approval of DOE's positions on these issues before proceeding with additional risk assessment activities under the new schedules for preparing primary RI/FS documents.

Examples of topics to be discussed include the models and equations used to estimate exposures; the numerical parameter values used in these models and equations, and assumptions affecting receptor location and reasonable maximum exposure (RME) scenarios. Other issues include the basis for selecting constituents of potential concern, the basis for selecting environmental transport and exposure pathways for quantitative evaluation, the methodology used to quantify the risks corresponding to the estimated exposures, the basis for identifying and selecting appropriate human receptors for quantification of RME scenarios, and the identification of critical subpopulations.

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Clearly defining the scope of risk assessment activities in the Work Plan Addendum is critical for the timely completion of the RI/FS at such a complex site. All parties involved, including the

DOE, EPA, contractors, and the State of Ohio, must have a common understanding of what is to be accomplished by the RI/FS risk assessment process for the FEMP.

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The ultimate goal of remediation of the site is to be protective of human health and the environment. This goal applies to the entire site. Because site remediation is being managed on the basis of operable units covering distinct portions of the site, it is critical to establish a mechanism for determining whether estimated risks associated with selected remedial alternatives for individual operable units are protective when considered collectively.

1.2 JUSTIFICATION FOR WORK PLAN ADDENDUM

The previously approved RI/FS Work Plan contains neither sufficient nor current descriptions of the risk assessment scope and methodology. It is insufficient because:

- New risk assessment guidance has become available since its approval.
- The risk assessment guidance inadequately addresses certain issues.
- The operable unit approach has been incorporated into the RI/FS process since the previous Work Plan was approved.

This addendum to the Work Plan includes new risk assessment guidance available to date and describes the technical approach to be used in the absence of guidance on specific, critical issues. This addendum describes operable unit and site-wide risk assessment activities that will be performed during the RI/FS.

1.3 ORGANIZATION OF WORK PLAN ADDENDUM

This Work Plan Addendum consists of ten sections - distinct, but closely related. Section 1.0 includes discussion of the intent and justification for an addendum to the work plan, the organization of the addendum, an introduction to the operational history at the site, an introduction to the RI/FS process at the site, and an introduction to plans for completion of the RI/FS at the site.

Section 2.0 presents the strategy for completing risk assessment tasks for the RI/FS. The section also presents the relative sequence and interrelationships of risk assessment tasks and deliverables. In addition, risk assessment concerns are addressed from an operable unit and a site-wide perspective.

Section 3.0 initiates the discussion of the risk assessment process itself and briefly addresses sources of information and analytical data to be used in the risk assessments for the FEMP RI/FS. Section 4.0 proceeds with a discussion of contaminants of potential concern for the risk assessment. Section 5.0 addresses development of exposure scenarios. Section 6.0 presents a discussion of the fate and transport modeling used in the risk assessment process for the FEMP. Section 7.0 presents the methodology for quantification of intakes for exposure scenarios previously developed in Section 5.0. Toxicity assessment for contaminants of potential concern is addressed in Section 8.0. Section 9.0 presents methodology for characterization of risks associated with the intakes quantified in Section 7.0. A strategy for simultaneously managing risks on an operable unit and a site-wide basis is presented in Section 10.0. The risk assessment process is also summarized in Section 10.0 in terms of the results of risk assessment and their significance in the RI/FS process and the risk management decision-making process for the FEMP.

1.4 HISTORY OF THE SITE

The FEMP is a government-owned, contractor-operated federal facility, which produced pure uranium metals for DOE. The FMPC began operations at the Fernald site, located in southwestern Ohio in the early 1950s as part of a long-term plan by the United States Atomic Energy Commission (AEC) to establish an integrated in-house uranium processing production complex. The entire site was operational by the end of 1954. In 1951, NLO, Inc. (formerly National Lead Company of Ohio), a subsidiary of NL Industries (formerly the National Lead Company), New York, entered into contract with the DOE (formerly the AEC) as operator of the FMPC. NLO, Inc. continued as the FMPC contract operator until January 1, 1986, when the Westinghouse Environmental Management Company of Ohio (WEMCO) (formerly Westinghouse Materials Company of Ohio [WMCO]), a wholly-owned subsidiary of the Westinghouse Electric Corporation, began contract responsibilities for management of the site operations and facilities for a five-year period. In 1991, DOE renamed the site the FEMP. WEMCO continues to operate the FEMP for DOE, with a contract extension through March 1992.

The FMPC utilized a wide variety of chemical and metallurgical processes to produce uranium metals. These operations were generally confined to specific areas of the site. The FMPC converted both uranium ore concentrates and "recycle materials" into high purity uranium metal having several standard isotopic assays. The isotopic values ranged up to 1.4 percent uranium-235 (U-235) by weight of the total uranium content of the product. However, most of the metal produced by the FMPC was depleted uranium. This metal was cast into ingots and shipped to the DOE facilities located at Reactive Metals, Incorporated (RMI), Ashtabula, Ohio, for extrusion into bars. Some of the extrusions were returned to the FMPC for heat treating and fabrication into target element cores for DOE reactors. Production peaked in 1960 at approximately 10,000 metric tons (MT) of uranium per year. A production decline began in 1964 and reached a low of 1230 MT per year in 1975. Production increased again in the early 1980s, and all production ceased in the summer of 1989.

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In addition to uranium foundry operations, the FMPC processed small amounts of thorium during the period 1954 through 1975. These operations were performed in the metals fabrication plant, recovery plant, special products plant, and the pilot plant. Since 1975, the FMPC has received, assayed, and stored quantities of thorium-bearing materials for potential use in future DOE programs. The site maintains long-term storage facilities for a variety of thorium materials as part of its role as the thorium repository for DOE.

Additional information on the history of the FMPC is included in the RI/FS Work Plan (DOE 1988a) and subsequent RI/FS reports.

1.5 PHYSICAL DESCRIPTION OF THE SITE

The FEMP property houses an inactive industrial site on 1050 acres in Hamilton and Butler counties, approximately 20 miles northwest of Cincinnati, Ohio (Figure 1-1). Bounded on the west and south sides by roads, the perimeter of the irregularly shaped property is completely fenced, with the exception of two road entrance portals. A second inner fence line surrounds the production area and waste disposal area. The facility contains several large buildings made of a variety of materials including concrete, brick, metal, and wood, as well as several waste ponds and storage silos. The structures contain stored materials and inactive process equipment. A railroad spur runs along the north side of the production and waste disposal areas. There are currently no residences on the FEMP property.

Situated on relatively flat terrain, the FEMP property slopes gently from the northeast to the southwest. The property is generally open grassland, with wooded areas on its southern, western, and northern portions. The primary topographic feature on the property is a gully containing Paddys Run, an intermittent stream located to the west of the production area and waste storage area. A small tributary of Paddys Run known as the Storm Sewer Outfall Ditch is located to the south and east of the production area.

Additional descriptions of the site and its environs are found in the RI/FS Work Plan (DOE 1988a) and subsequent RI/FS reports.

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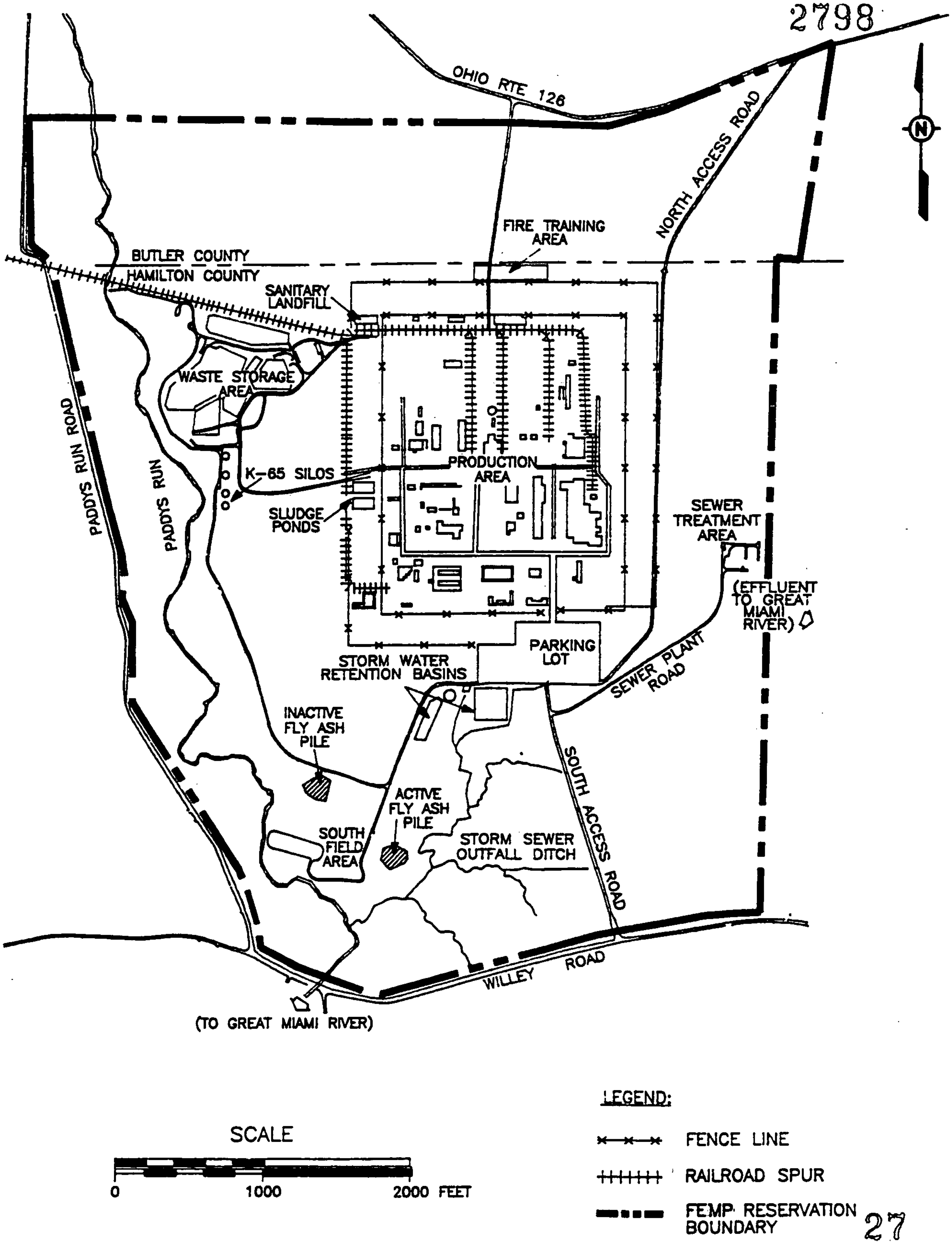

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FIGURE 1-1 FERNALD ENVIRONMENTAL MANAGEMENT PROJECT

1.6 RI/FS ACTIVITIES

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Work performed on the RI/FS to date has provided extensive characterization of environmental transport and contaminant distribution patterns in the regional aquifer, distribution patterns of contaminants in soils on and surrounding the FEMP, and a preliminary indication of contaminant inventories and distributions in waste areas that constitute potential sources of contamination to the environment. Supplemental field investigation studies are in progress or are planned, which will complete the site characterization process. Results from these studies are needed before operable unit and site-wide RI/FS reports can be finalized; however, work on many RI/FS report tasks are continuing while additional field investigation studies are being conducted.

Work performed on the RI/FS process has led to the development of an understanding of the site that is crucial to completion of the RI/FS. The planned approach for completion of the RI/FS maximizes the use of previous operable unit RI/FS resources and documents. Key features of the plan for completion of the RI/FS process at the site include:

- Continue with the operable unit approach in the RI and FS processes.
- Revise the definitions of operable units.
- Address site-wide risk concerns by supplementing the operable unit approach with a Preliminary Site-Wide Baseline Risk Assessment and FS Comprehensive Response Action Risk Evaluations.
- Apportion site-wide risk limits to operable units through an iterative mechanism implemented in parallel with the operable unit FS processes. This is intended to provide a mechanism for developing and refining remediation goals.

Continuation of the operable unit approach includes generation of primary RI and FS reports for each operable unit. The RI report for each operable unit will contain a baseline risk assessment. The FS report for each operable unit will contain risk assessments for each remedial alternative. In addition, an FS Comprehensive Response Action Risk Evaluation will be included in the FS report for each operable unit. This site-wide risk assessment will address the cumulative protectiveness of selected operable unit remedial alternatives for the entire site.

Continuation of the operable unit approach will be accomplished within the framework of revised operable unit definitions. The most technically and programmatically meaningful definitions of operable units have evolved as a result of insight gained during RI and FS activities conducted to date. Although some rework of previous RI/FS efforts will be necessary as a result of the redefinition, it is intended that the revised definitions for operable units facilitate the overall

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completion of the RI/FS at the FEMP. The revised operable unit definitions are addressed in Section 1.7.

The operable unit FS risk assessments will be supplemented with FS Comprehensive Response Action Risk Evaluations in order to ensure that estimated risks associated with remediation are protective of human health and the environment when the site is considered as a whole. The comprehensive evaluation will be revised to accommodate changes in the remedial alternatives for the site as the preferred alternative is selected in the FS for each operable unit. Iterations of this site-wide assessment task will reveal the contribution of individual operable units to site-wide risks. This information will be used to determine the portion of the site-wide risk limit that may be allotted to each operable unit and ultimately to each pathway and contaminant of concern for each operable unit. Apportionment of site-wide risks will facilitate derivation of cleanup levels for contaminants of potential concern for each operable unit.

1.7 OPERABLE UNIT DEFINITIONS

Operable unit definitions for the RI/FS at the FEMP have been revised. The operable unit definitions listed in this Work Plan Addendum are made to comply with the requirements in the Amended Consent Agreement. Operable Units 1 through 5 are shown in Figures 1-2 through 1-4. In Figure 1-2, state planar coordinates for Operable Units 1, 2, and 4 are tabulated and these boundaries are illustrated on the site map. The definitions of Operable Units 3 and 5 are noted at the bottom of Figure 1-2. The revised definitions are presented below:

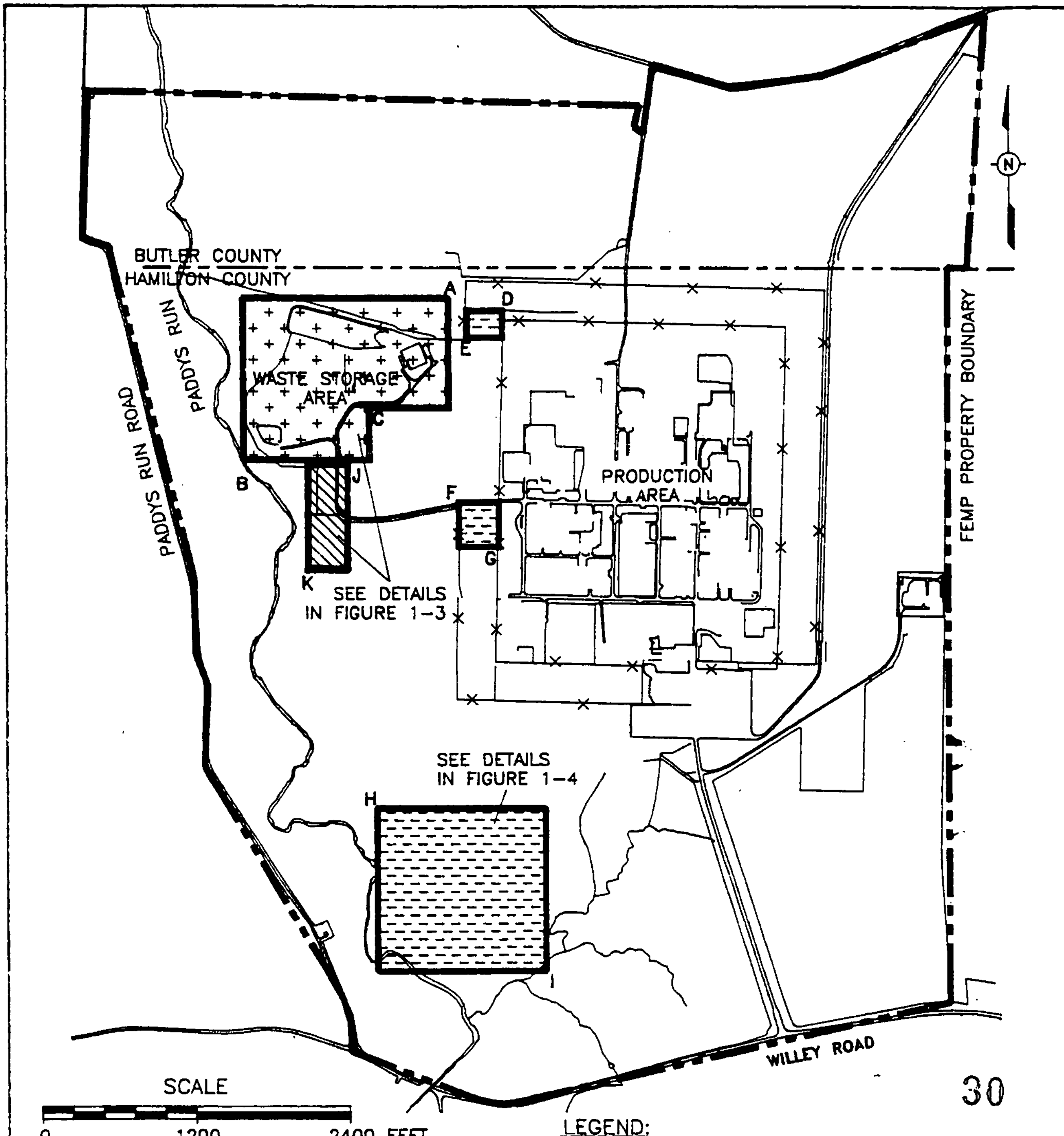
- Operable Unit 1 is defined as Waste Pits 1 through 6, the Clearwell, the Burn Pit, berms, liners, and soil within the operable unit boundary (Figures 1-2 and 1-3).

- Operable Unit 2 is defined as the fly ash piles, other Southfield disposal areas, the lime sludge ponds, the solid waste landfill, berms, liners, and soil within the operable unit boundary (Figures 1-2, 1-3, and 1-4).

- Operable Unit 3 is defined as the production area and production-associated facilities and equipment (includes all above- and below-grade improvements) including, but not limited to, all structures, equipment, utilities, drums, tanks, solid waste, waste, product, thorium, effluent lines, K-65 transfer line, wastewater treatment facilities, fire training facilities, scrap metal piles, feedstocks, and the coal pile.

- Operable Unit 4 is defined as Silos 1, 2, 3, and 4, berms, the decant tank system, and soil within the operable unit boundary (Figures 1-2 and 1-3).

- Operable Unit 5 is defined as groundwater, soil not included in the definitions of Operable Units 1, 2, 3, and 4, surface water, sediments, flora, and fauna.



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NOTE:

OPERABLE UNIT 3 INCLUDES ALL BUILDINGS, PIPELINES, AND ABOVEGROUND STRUCTURES IN THE PRODUCTION AREA. OPERABLE UNIT 5 INCLUDES GROUNDWATER, SURFACE WATER, SOILS, FLORA AND FAUNA IN THE REGIONAL AREA AS WELL AS THE PRODUCTION AREA.

LEGEND:

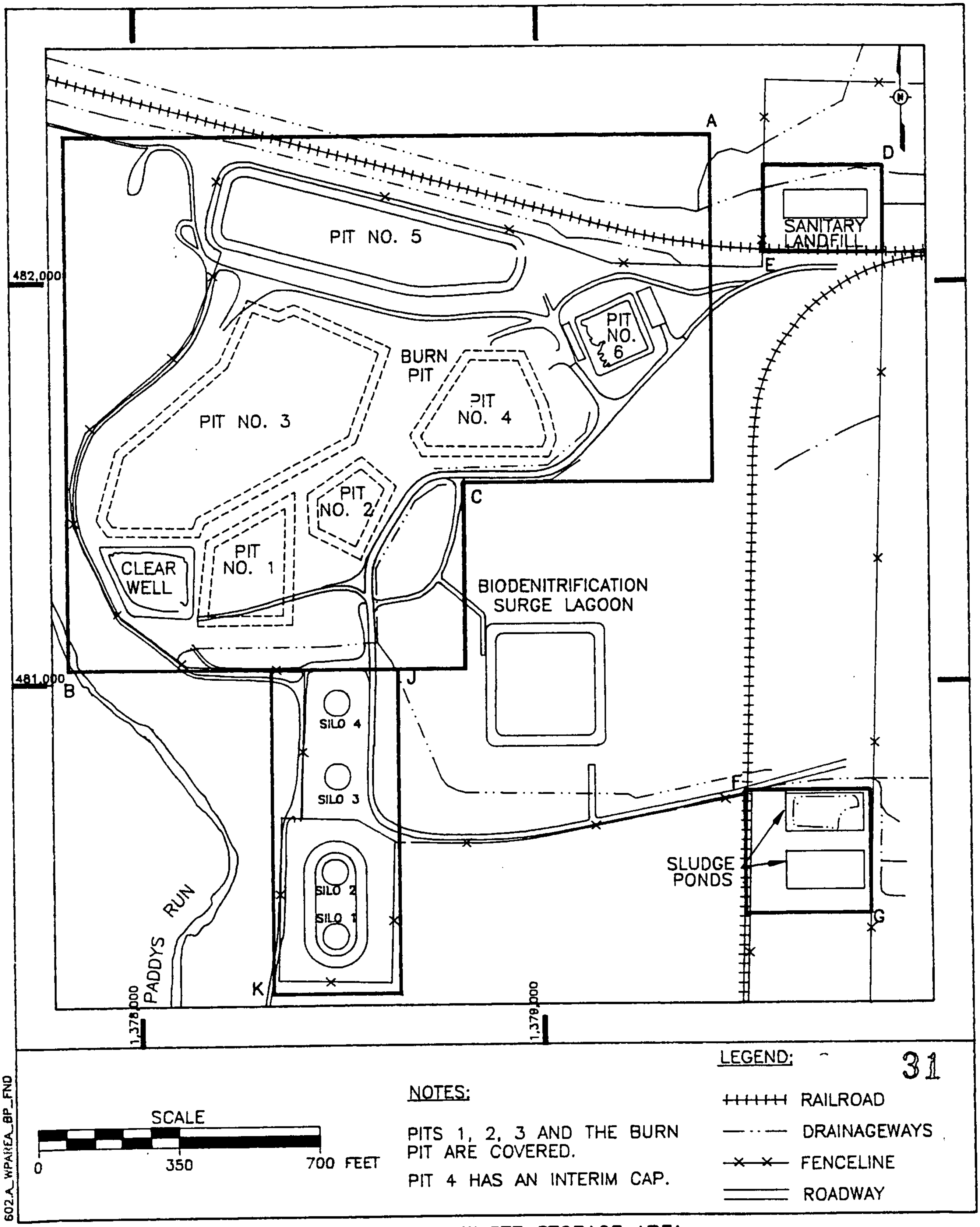
- OPERABLE UNIT 1
- OPERABLE UNIT 2
- OPERABLE UNIT 4

STATE PLANAR COORDINATES

A	N482364	E1379432
B	N481033	E1377824
C	N481499	E1378812
D	N482264	E1379849
E	N482051	E1379568
F	N480756	E1379501
G	N480402	E1379819
H	SEE FIGURE 1-4	
I	SEE FIGURE 1-4	
J	N481033	E1378642
K	N480222	E1378329

KNOX/RA-WP/AB.5-5/02-04-92

FIGURE 1-2. RI/FS OPERABLE UNITS



NOTES:

PITS 1, 2, 3 AND THE BURN PIT ARE COVERED.

PIT 4 HAS AN INTERIM CAP.

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FIGURE 1-3. WASTE STORAGE AREA

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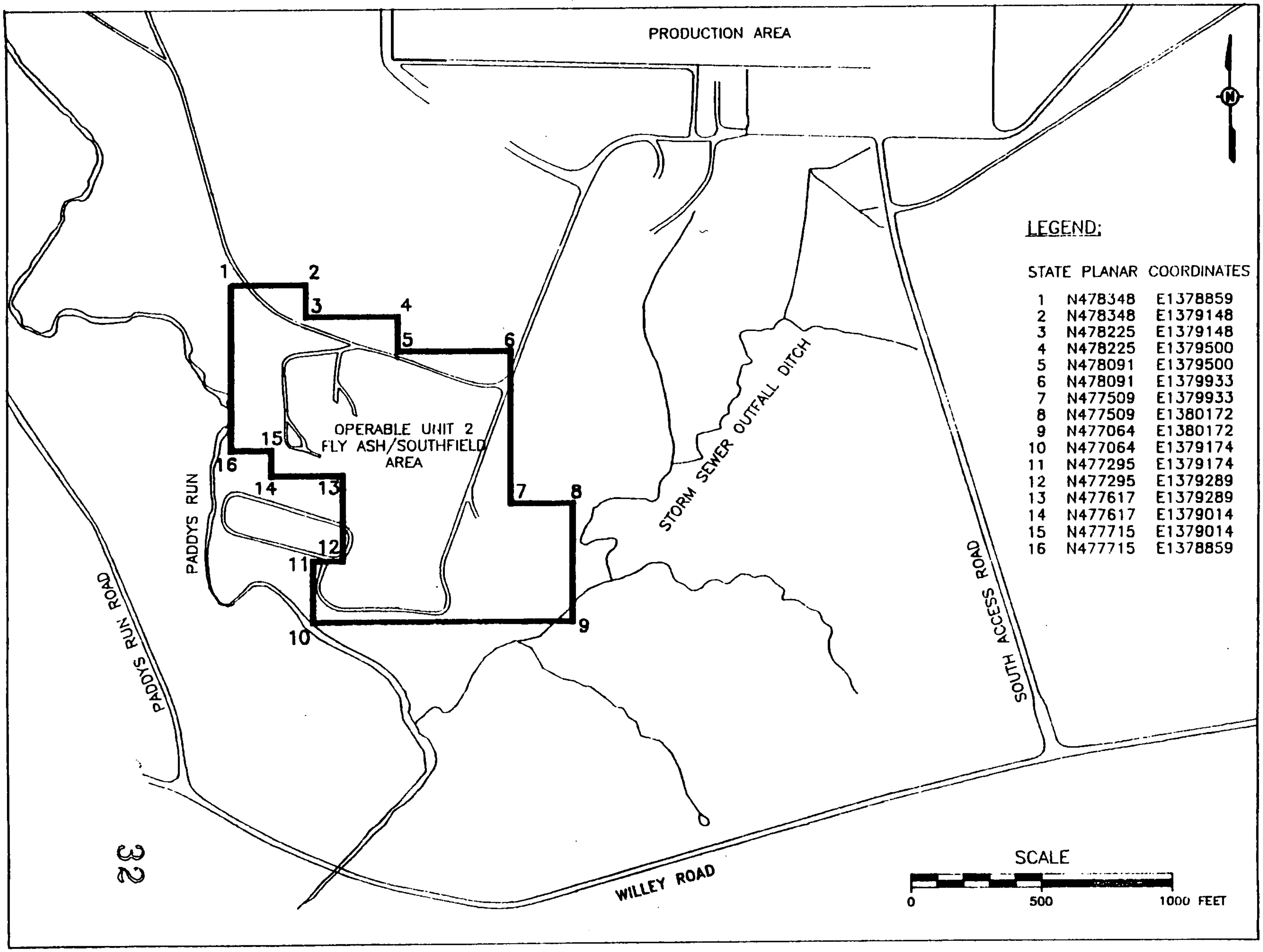

LEGEND:

STATE PLANAR COORDINATES		
1	N478348	E1378859
2	N478348	E1379148
3	N478225	E1379148
4	N478225	E1379500
5	N478091	E1379500
6	N478091	E1379933
7	N477509	E1379933
8	N477509	E1380172
9	N477064	E1380172
10	N477064	E1379174
11	N477295	E1379174
12	N477295	E1379289
13	N477617	E1379289
14	N477617	E1379014
15	N477715	E1379014
16	N477715	E1378859

FIGURE 1-4. OPERABLE UNIT 2, FLY ASH/SOUTHFIELD AREA

Evaluation of contaminated groundwater-related risk and treatment technologies is to be considered in Operable Unit 5, except as required under removal actions for other operable units.

- The Comprehensive Site-Wide Operable Unit represents the entire site and is defined as an operable unit for the purpose of evaluating the remedies selected for the five operable units (including remedial and removal response actions) to ensure that they are protective of human health and the environment on a site-wide basis as required by CERCLA, the National Oil and Hazardous Substances Pollution Contingency Plan (NCP) (EPA 1990a), and applicable U.S. EPA policy and guidance.

The definitions of Operable Units 1 through 4 each include water encountered during response actions associated with those operable units.

2.0

2.0 RISK ASSESSMENT PROGRAMMATIC APPROACH

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This section of the work plan describes the overall objectives of a risk assessment and the specific objectives of a baseline and an FS risk assessment. The objectives of the site-specific baseline and FS risk assessments for the individual operable units and for the entire site are discussed in Sections 2.2 and 2.3, respectively. The Site-Wide Characterization Report is briefly discussed in relation to the risk assessment process in Section 2.4. The technical approach for integrating the site-specific risk assessments is presented in Section 2.5. The format for presentation of the site-specific risk assessments is described in Section 2.6.

2.1 OBJECTIVES OF RISK ASSESSMENTS

The mandate of the Superfund Comprehensive Environmental Response, Compensation, and Liability Act of 1980 (CERCLA) program is to protect human health and the environment from current and potential threats posed by uncontrolled hazardous substance releases. The potential threat to human health and the environment is evaluated and documented via the risk assessment process. The goal of the risk assessment process is to provide risk information necessary to assist decision-making at remedial sites. This risk information is developed in the baseline risk assessment during the RI process and in the risk assessment for remedial alternatives during the FS process. The objectives of the baseline and FS risk assessments are discussed below.

2.1.1 Objectives of a Baseline Risk Assessment

The objective of a baseline risk assessment is to evaluate and document the potential risks to human health and the environment associated with current and predicted future exposures to site-related contaminants if no remedial action is taken. This information provides a basis for determining whether remediation is necessary at the site. The risks determined in the baseline risk assessment represent the risk for the no-action alternative in the FS risk assessment. In addition, the baseline risk assessment provides a basis from which, during the FS, acceptable levels of contaminants that can remain on site are determined.

The process used to accomplish the objectives of a baseline risk assessment is summarized in Figure 2-1. The following tasks are conducted in a baseline risk assessment:

- Identify all radionuclides and chemicals of potential concern at the site.
- Conduct exposure assessments for site-related radionuclides and chemicals of potential concern.

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- Assess the toxicity of site-related radionuclides and chemicals of potential concern.

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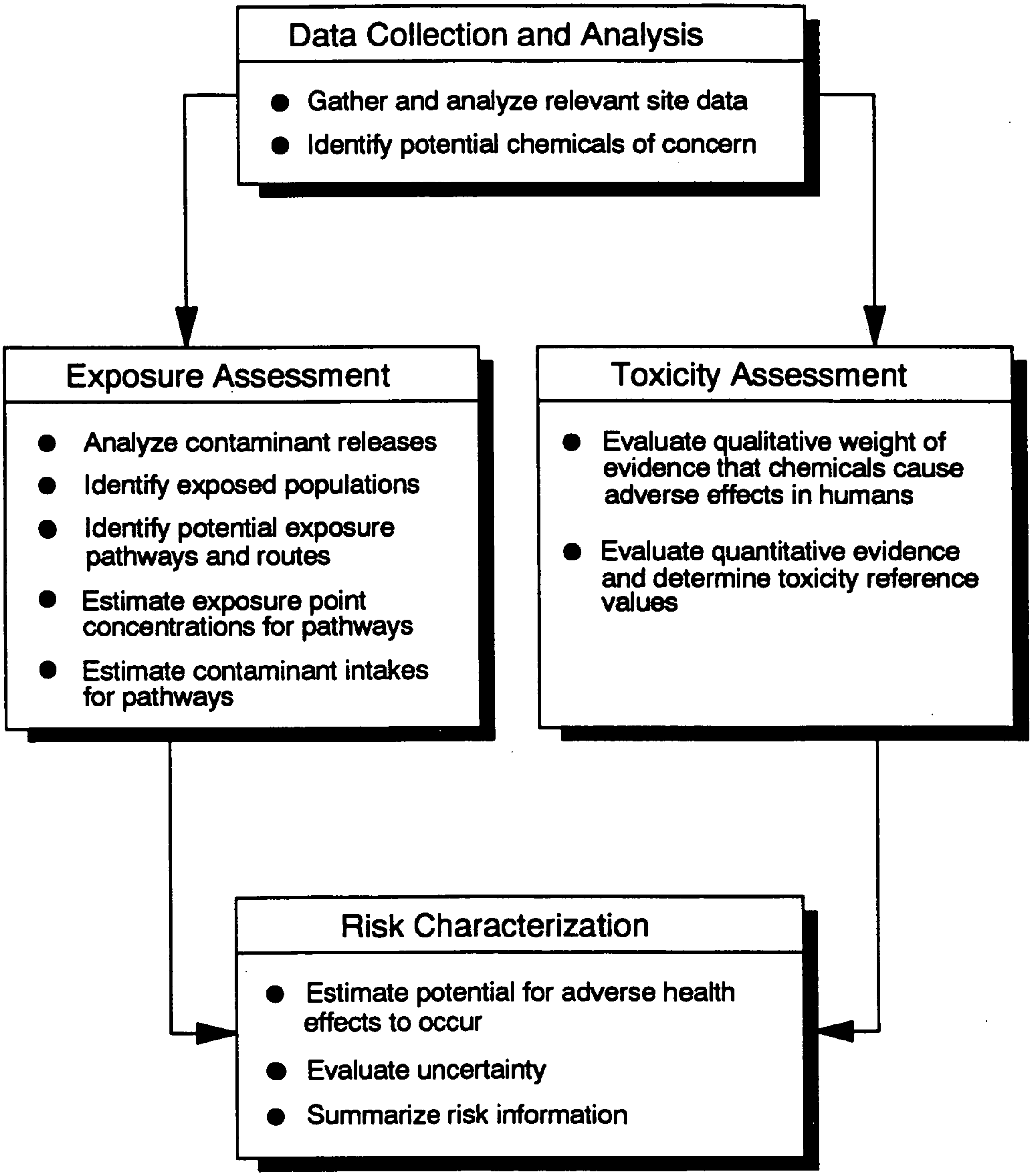

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Source: Adapted from EPA, 1989a

FIGURE 2-1

BASELINE RISK ASSESSMENT PROCESS

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- Quantify risks to human health.
- Quantify risks to ecological receptors.

In addition, a baseline risk assessment should provide recommendations, as necessary, for supplemental investigations of the site and should support the development of preliminary remediation goals, final remediation goals, and remedial action objectives.

2.1.2 Objectives of an FS Risk Assessment

Each proposed remedial alternative considered in an FS has various benefits and risks associated with it. The objective of the risk assessment portion of an FS is to evaluate and document the types and magnitudes of potential adverse impacts on human health and the environment from each remedial alternative. This evaluation must provide an assessment of the long-term effectiveness and permanence of each alternative for reducing the magnitude of residual risks present after remediation. Additionally, the FS risk assessment must assess the short-term effectiveness of the alternative to protect the community, the workers, and the environment during remediation. The results of the FS risk assessment must be presented in a form that allows for the following:

- Evaluation of the overall protectiveness of the alternatives
- Comparison of the risks for the different alternatives
- Determination of the degree to which preliminary and final remediation goals and remedial action objectives are met

2.2 OPERABLE UNIT RISK ASSESSMENTS

Operable unit risk assessments deal with those risks to human health and the environment which are associated with the individual operable units at the FEMP and any remedial action alternatives for those operable units.

2.2.1 Operable Unit Baseline Risk Assessments

A baseline risk assessment will be performed on each operable unit. Each baseline risk assessment will compile and evaluate all pertinent information currently available for that operable unit. These operable unit databases will be compiled from the data sources listed in Section 3.0. Each operable unit database will provide the information needed to:

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- Characterize the source(s) associated with that operable unit.
- Determine the contaminants of concern for that operable unit.

- Identify the significant exposure pathways for that operable unit.
- Assess contaminant transport from that operable unit over the next 1000 years.
- Quantify significant exposures attributable to the operable unit.
- Select the RME scenario for the operable unit.

Risks associated with the operable unit will be assessed for the RME scenario assuming no remediation. Credit will not be taken for removal actions within an operable unit unless the removal action has been completed at the time of the operable unit baseline risk assessment. Agency decision-makers will review the calculated baseline risks to determine if the configuration of the operable unit is sufficiently protective of human health and the environment, both now and in the future, if no action is taken. If it is determined that human health and the environment are not sufficiently protected, remedial alternatives will be developed and the baseline risk will be compared with the risks associated with the remedial alternatives.

The baseline risk assessment will provide documentation on the methodology used to determine the risks from the operable unit. It will also clearly present the resulting estimated doses and risks associated with the baseline scenario.

2.2.2 <u>Operable Unit FS Risk Assessments</u>

During the detailed analysis of alternatives phase of the FS process, various remedial alternatives will be evaluated with respect to a specific list of criteria, including the criteria listed in Section 2.1.2. The risk assessment portions of the FS process involve the identification and quantification of risks associated with each alternative considered. Each operable unit FS risk assessment will:

- Calculate and present the estimated short- and long-term risks associated with each proposed FS alternative.

- Provide input into the FS Comprehensive Response Action Risk Evaluation (Section 2.3.2).

- Summarize the results of the above tasks and document both the methodology and data sources used to perform them.

The FS risk assessment will provide a documented estimate of the human health and ecological risks associated with each remedial alternative; and will be used by decision makers in the overall evaluation of alternatives in the FS process.

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2.3 SITE-WIDE ASSESSMENTS

This group of assessments deals with those risks to human health and the environment which are associated with the FEMP as a whole.

2.3.1 Preliminary Site-Wide Baseline Risk Assessment

The Preliminary Site-Wide Baseline Risk Assessment, a part of the Site-Wide Characterization Report (Section 2.4), will yield a site-wide perspective of risks under current conditions and predicted future scenarios if no action is taken. The Preliminary Site-Wide Baseline Risk Assessment will present all pertinent information available as of December 1, 1991 on the five operable units, as well as for the whole site. The data for the Preliminary Site-Wide Baseline Risk Assessment will be compiled from the sources listed in Section 3.0. These data will be evaluated as part of the Preliminary Site-Wide Baseline Risk Assessment to:

- Characterize all potential sources of contaminant release to the environment.
- Determine the contaminants of potential concern for the site.
- Identify the pathways capable of producing significant exposures from the site.
- Assess contaminant transport within or from the site over the next 1000 years.
- Quantify significant exposures.
- Quantify contaminant- and pathway-specific risks and combine comparable human health risks from multiple contaminants to common receptors.
- Select the RME scenarios for the FEMP.

Risks associated with contaminants at the FEMP will be assessed for the RME scenarios assuming no remediation. Evaluation of operable unit baseline risks and baseline risks for the entire FEMP will:

- Provide information needed to determine if current or future conditions at the FEMP are sufficiently protective of human health and the environment on a comprehensive basis.
- Identify and rank individual sources, contaminants, and pathways contributing to the total risk from the site.
- Provide a basis for prioritizing further removal actions.
- Support development of site-wide preliminary remediation goals.

- Provide the risk estimates for the "no-action" alternative in the Comprehensive Response Action Risk Evaluation (Section 2.3.2) in the operable unit FS.

The Preliminary Site-Wide Baseline Risk Assessment will provide documentation on the methodology used to perform all tasks required to quantify the risks from the site. It will present the relevant results and conclusions of previous RI/FS documents.

2.3.2 FS Comprehensive Response Action Risk Evaluations

Each operable unit remedial alternative has some degree of long-term and short-term risk associated with it. For example, it is likely that each operable unit alternative will have some level of long-term risk associated with it. Although the intention of many of the proposed remedial alternatives at the FEMP is to remove, isolate, or immobilize contaminants, these remedial actions may leave traces of mobile contaminants or "residuals" on site. The potential risks to future receptors from these residuals will be known as "residual risks" throughout this RI/FS process. The combined residual risks from all operable units must be evaluated to ascertain if their aggregate residual risks remain protective of human health and the environment.

The activities associated with each remedial alternative are expected to generate short-term risks to remediation workers and the public. The magnitude of these risks and their target populations must be assessed to determine if these risks (i.e., transportation, construction accidents, exposures, etc.) are sufficiently protective of human health when combined with similar risks to the same receptors from other operable units.

The FS Comprehensive Response Action Risk Evaluation provides the mechanism to assess the cumulative impact of risks associated with each operable unit's remediation. As part of the FS process for each operable unit, the level of residual risk will be estimated for each remedial alternative considered for that unit. The remaining risks from the most likely configuration of the other operable units, after their remediation, also will be determined. To do this, the remedial alternative most likely to be implemented for each operable unit must first be determined. If an operable unit has successfully completed the FS portion of the RI/FS process, the selected alternative and accompanying risk estimates will be used to assess its site-wide impacts. If the operable unit has not completed the FS process, then a surrogate FS alternative, known hereafter as the "Leading Remedial Alternative," and an estimate of its risks will be used. The Leading Remedial Alternative for each operable unit will be identified and presented in the Site-Wide Characterization Report (Section 2.4). The Leading Remedial Alternative does not represent the pre-selection of a remedy and will be used only for the purpose of estimating and evaluating the risks presented by the entire site during the FS/Comprehensive Response Action Risk Evaluation for Operable Units 1 through 5.

Contaminant- and pathway-specific short-term and residual risks will be quantified for each operable unit Leading Remedial Alternative. The resultant operable unit residual risks then will be summed to estimate the short-term and residual risks attributable to the FEMP as a whole. Thus, the cumulative long-term (i.e., residual) and short-term risks corresponding to the selected or surrogate alternative for every operable unit will be evaluated on a progressive basis during the course of each individual operable unit FS.

2.3.3 Site-Wide Projected Residual Risk Assessment

The Site-Wide Projected Residual Risk Assessment will present an assessment of site-wide risks that are anticipated to remain at the FEMP following implementation of the selected response actions embodied in the Records of Decision (RODs) for Operable Units 1 through 5 and the selected removal actions. The Site-Wide Projected Residual Risk Assessment will be based on site-specific measurements included in the supporting documents for the RODs for Operable Units 1 through 5 and supplemented by environmental transport modeling results for future hypothetical exposure scenarios. The assessment will:

- Include previous fate and transport, and exposure modeling results produced for the operable unit baseline and FS risk assessments, where appropriate.
- Provide a comprehensive assessment of potential risks associated with remedial alternatives actually selected for all portions of the site.
- Present and incorporate any additional FEMP characterization data not in any earlier report.
- Refine the estimate of impacts of locating an on-site waste management facility once all anticipated waste volumes, types, and forms are known, if such a facility is part of a remedial alternative.
- Identify significant remaining sources of residual risks.
- Establish the basis for additional actions if the final planned combination of operable unit remedial actions produces residual risks that are generally not protective of human health and the environment.

2.3.4 Site-Wide Feasibility Study Risk Assessment

A Site-Wide Feasibility Study of additional remedial action alternatives will be necessary only if the residual risks from the FEMP, as determined by the Site-Wide Projected Residual Risk 40

Assessment, are not considered to be protective of human health and the environment. This task provides a mechanism that will ensure the final combination of FS remedial alternatives will produce a site-wide residual risk that is protective of human health and the environment. This assessment will:

- Include the Site-Wide Projected Residual Risk Assessment as the no-action alternative.
- Provide a comprehensive assessment of potential risks associated with any additional remedial alternatives proposed for the site.
- Address the impacts of placing any additional waste in an on-site waste management facility.
- Document that the final planned combination of operable unit remedial actions and additional actions will produce residual risks that are generally protective of human health and the environment.

2.4 SITE-WIDE CHARACTERIZATION REPORT

Data pertaining to the site conditions as of early 1988 were assembled by DOE as part of the RI/FS Work Plan process. Since that time, a considerable amount of new information on the potential sources of contaminants and the nature and extent of environmental contamination at the site has been generated through the RI for the operable units and through other environmental programs at the FEMP. Although much of this information has been compiled and presented in reports for individual operable units, there has not been a presentation of all data to characterize the entire site and under the previous Consent Agreement schedules the only RI report delivered to EPA was for Operable Unit 4.

In order to bring together characterization data for the entire site and to support the operable unit and site-wide RI/FS activities, a Site-Wide Characterization Report will be prepared. This report will provide a one-time summary of all site data available as of December 1, 1991. The report will also contain a Preliminary Site-Wide Baseline Risk Assessment (Section 2.3.1) that characterizes the current and potential threats to human health and the environment that may be posed by contaminants at the entire site.

Based on the data presented in the Site-Wide Characterization Report and on best professional judgement, the Leading Remedial Alternatives for Operable Units 1 through 5 will be identified and presented in the report. The Leading Remedial Alternative for each operable unit is the remedial alternative considered most likely to be selected as the preferred alternative for that operable unit. As stated previously, it does not represent the pre-selection of a remedy but will

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be used only to estimate and evaluate the risks presented by the entire site within the Comprehensive Response Action Risk Evaluations of the Operable Unit FS reports (Section 2.3.2). The Leading Remedial Alternative will in no way prescribe or restrict the selection of the remedy for Operable Unit 1 through 5 RODs.

2.5 RISK ASSESSMENT TECHNICAL APPROACH

The overall risk assessment technical approach is developed within the context of the entire RI/FS process for the FEMP. The DOE will complete the RI/FS for the FEMP by implementing the RI and FS processes for each operable unit of the site. Consistent with the operable unit approach, an ROD will be prepared at the end of each operable unit RI/FS. In addition, an ROD for the entire site will be issued following the determination that the selected alternatives for each operable unit are protective of human health and the environment when considered either individually or collectively. Therefore, the risk assessment technical approach is predicated on completion of the RI/FS process based on the operable unit concept. This technical approach is presented conceptually in Figure 2-2. The figure identifies specific RI and FS risk assessment tasks for each operable unit at the FEMP. It also identifies other RI/FS tasks and interactions among these tasks and the risk assessment tasks.

Within the context of the operable unit technical approach, the mechanism for evaluating protection of human health and the environment from the entire site is dependent on inclusion of an FS Comprehensive Response Action Risk Evaluation appended to each operable unit FS report. These site-wide assessments will be based on the selected remedial alternative from each operable unit FS or a Leading Remedial Alternative from each operable unit FS that has not completed the selection process. Since the operable unit FS processes will not be synchronized, the FS Comprehensive Response Action Risk Evaluations will be iterative, reflecting selection of an alternative for a particular operable unit as its FS schedule nears completion. This iterative mechanism will provide estimates of site-wide risks associated with remediation of the entire site beginning at an early stage in the RI/FS process. The iterations will then undergo refinement through later stages of the RI/FS process.

The results of the FS Comprehensive Response Action Risk Evaluations will reveal whether proposed remedial actions at a given operable unit will afford protection when integrated into the site-wide strategy. If overall protection is not indicated, remedial alternatives must be re-examined to determine what changes might be made to one or more operable unit remedial alternatives to achieve overall protection from the site.

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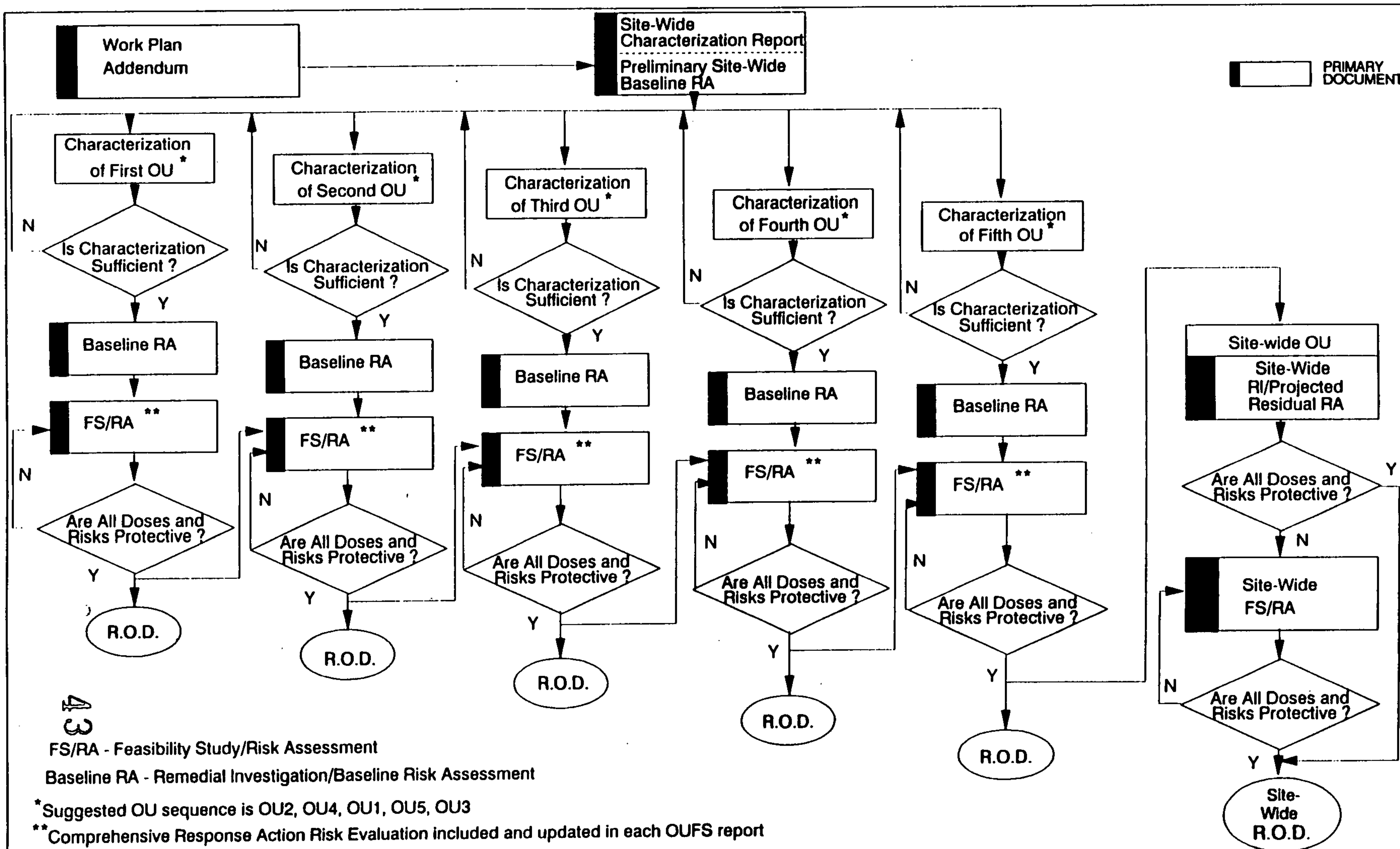

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FS/RA - Feasibility Study/Risk Assessment

Baseline RA - Remedial Investigation/Baseline Risk Assessment

*Suggested OU sequence is OU2, OU4, OU1, OU5, OU3

**Comprehensive Response Action Risk Evaluation included and updated in each OUFS report

Figure 2-2. Conceptual Representation of the Information Flow for the RI/FS Risk Assessment Process

The technical approach facilitates timely performance of RI/FS tasks. The operable unit technical approach accommodates initiation of operable unit RI and FS tasks based on work that has been performed to date. Results generated from planned and ongoing field investigations that will complete the site characterization effort will be systematically incorporated into the process as they become available. Complete characterization of an operable unit is only required before the risk assessments for that operable unit are finalized.

2.6 PRESENTATION OF RISK ASSESSMENTS

This section addresses the presentation format for RI and FS risk assessment reports and identifies the risk assessment reports that will be generated. The discussion in this section addresses baseline and FS risk assessments for operable units and a Site-Wide RI/Projected Residual Risk Assessment report following completion of operable unit reports.

2.6.1 General Risk Assessment Report Format

2.6.1.1 Baseline Risk Assessment Format

The EPA provides detailed guidance concerning the format of the baseline risk assessment report. This guidance is presented in the Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual, (Part A) (EPA 1989a). This guidance document is a source of baseline risk assessment methodology as well as report format guidance. The suggested outline for a baseline risk assessment report is included in the EPA guidance document and is reproduced in Attachment I of this addendum. This outline forms the basis for the format to be used in the RI/FS baseline risk assessments. The suggested EPA outline will be modified, however, to accommodate assessment of ecological impacts and complement the information presented in the RI report.

2.6.1.2 FS Risk Assessment Format

The EPA does not provide guidance concerning a format or methodology for FS risk assessments. The EPA guidance for conducting the RI/FS under CERCLA (EPA 1988a) only specifies the criteria that must be used to evaluate remedial alternatives. The FS risk assessment format adopted for the FEMP will address risk within the context of the evaluation criteria specified by EPA.

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2.6.2 Operable Unit RI/FS Risk Assessments

2.6.2.1 RI Baseline Risk Assessments

The risk assessment for the RI will be conducted for each operable unit. Complete details of the baseline risk assessment will be appended to each RI report in a format consistent with EPA guidance. The salient features and results of the baseline risk assessment will also be reiterated and summarized in the text of the RI report. Section 6.0 of the RI report will present a summary of the baseline risk assessment. Each baseline risk assessment will only address concerns related to that particular operable unit.

2.6.2.2 FS Risk Assessments

The risk assessments for the FS tasks will be conducted for each operable unit remedial alternative. These FS risk assessments will be appended to each FS report. The salient features and results of the FS risk assessments will also be discussed in those sections of the FS report that present evaluations of each remedial alternative with respect to the evaluation criteria specified by EPA. An FS Comprehensive Response Action Risk Evaluation will be appended to each operable unit FS report.

2.6.3 Site-Wide RI/Projected Residual Risk Assessment

The Site-Wide RI/Projected Residual Risk Assessment will present an evaluation of the combined risks from all contaminants and exposure pathways of concern from the entire site to confirm the efficacy of previous risk management decisions for each operable unit and the entire site. The Site-Wide RI/Projected Residual Risk Assessment report will follow completion of operable unit reports and will be prepared as a stand-alone document consistent with the format employed for operable unit FS risk assessments.

3.0

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3.0 OVERVIEW OF INFORMATION AND DATA UTILIZED IN RI/FS RISK ASSESSMENTS

This section addresses the types and sources of data and other site-specific information used in RI/FS risk assessments. The types of data used in RI/FS risk assessments are categorized in this section as:

- Data that characterize the site
- Data used to model the fate and transport of constituents
- Data used to estimate exposures

Data obtained during the RI/FS process are evaluated via the quality assurance (QA) program. Project QA objectives ensure that:

- Scientific data will be of sufficient or greater quality to meet scientific and legal scrutiny.

- Data will be gathered or developed in accordance with procedures appropriate for the intended use of the data.

- Data will be of known or acceptable precision, accuracy, completeness, representativeness, and comparability as required for the FEMP.

The QA program governing data acquisition and use is documented in the Quality Assurance Project Plan (QAPP) and supporting procedures that direct quality-related activities. The QAPP governing QA practices to be implemented for the FEMP RI is Volume 5 of the Work Plan Requirements and is entitled "Quality Assurance Project Plan, Revision 3" (DOE 1988a). This document includes the data quality objectives, the requirements for work performance to meet these objectives, the means for verifying that the objectives have been met, and a discussion of the data validation process. The RI/FS QAPP cited will be followed until the RI/FS begins operation under the site-wide QAPP, which is currently under revision.

Data generated in the RI/FS process are given first consideration in risk assessments because these data are the most current and most reliable based on the RI/FS quality assurance/quality control (QA/QC) practices. Data generated in DOE litigation studies of 1986-7 of off-property soil, surface water, sediment, and groundwater will be considered next because of the strict QA/QC practices applied in anticipation of their use in litigation (IT 1986, IT 1987). Existing databases generated by WEMCO and its subcontractors in routine environmental monitoring and in the Characterization Investigation Study (Weston 1987) will be considered as secondary sources because the QA/QC procedures on these data are not as well documented. If primary and secondary data do not corroborate each other, this will be noted and addressed and the primary data will be used for quantitative risk assessment calculations. Secondary sources will only be used when primary sources do not contain the data sought. If a secondary data source is used, the source of the data will be clearly identified.

3.1 SITE CHARACTERIZATION DATA

Site characterization data will be presented in the RI report. These data will not be repeated completely in the baseline risk assessment, which is a part of the RI. These data will be summarized, as necessary, in the risk assessment report.

Site characterization data indicate the extent of contamination in the environment from the site. The extent of contamination in the environment is determined from examination of background concentrations and constituent concentrations that can be attributed to releases from the site. Background levels of chemicals and radionuclides include naturally-occurring levels and concentrations that are present in the environment due to human-made, non-site sources (EPA 1989a). These data are obtained from a variety of sources such as, but not limited to, the sources of background data presented in Table 3-1. Data from these sources are used in RI/FS risk assessments according to the following hierarchy:

- Data to be considered first: site-specific data obtained from the RI/FS database, including data collected during removal actions

- If data from site-specific sources are insufficient, a second group of data will be considered. This group includes: other site-specific data from sources such as the environmental monitoring annual reports, county soil surveys, and site-specific studies that complement the RI/FS characterization process (e.g., Characterization Investigation Study, Facemire ecological survey of the FMPC site [Facemire et al. 1990])

- If data from the first two groups are insufficient, a third group of data will be considered. This group includes: regional data obtained from state and local sources or peer reviewed literature (subject to EPA approval)

In the absence of knowledge of background data for a contaminant in a specific medium, a background level of zero will be assumed for the contaminant in the specific medium.

The RI/FS database also includes the results from a number of special studies conducted as part of the RI/FS which will support the ecological risk assessment. These are the following:

TABLE 3-1
SOURCES OF BACKGROUND LEVELS OF CHEMICALS AND RADIONUCLIDES

Medium	Constituents	Sources
Air	Chemical Radiological	WMCO Environmental Monitoring Annual Reports[a]
External Radiation Exposure	Photon-Emitting Radionuclides	WMCO Environmental Monitoring Annual Reports
Groundwater	Chemical Radiological	RCRA Groundwater Background Wells
Surface Water	Chemical Radiological	WMCO Environmental Monitoring Annual Reports
Sediment[b]	Chemical Radiological	Shacklette et al. 1984, (Indiana/Ohio data only), Myrick et al. 1983 (Indiana/Ohio data only)
Soil[b]	Chemical Radiological	Shacklette et al. 1984, (Indiana/Ohio data only), Myrick et al. 1983 (Indiana/Ohio data only)

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a Westinghouse Environmental Monitoring Annual Reports - WMCO 1986; WMCO 1987a; WMCO 1988; WMCO 1989; WMCO 1990.

b Site-specific sampling for soil background levels will be performed in accordance with the Background Sampling and Analysis Plan under review by EPA. Data obtained from this program will be used in all risk assessments performed following acquisition of these site-specific data. Chemicals and radionuclides for which background sampling and analysis will not be performed are assumed to have a background level of zero.

- Analyses of radionuclides and chemicals in plants, terrestrial animals, and aquatic organisms collected from the FEMP

- Surveys of macroinvertebrate communities in Paddys Run and the Great Miami River

- Toxicity tests of FEMP effluents

- Delineation of jurisdictional wetlands on FEMP property

- Toxicity tests of soil and sediment samples from the FEMP

As described in Section 2.4, the Site-Wide Characterization Report will provide a comprehensive summary of site characterization data available for RI/FS risk assessments as of December 1, 1991. The Site-Wide Characterization Report will incorporate and support the development of the Preliminary Site-Wide Baseline Risk Assessment. Information from the Site-Wide Characterization Report, supplemented with results of scheduled sampling and analysis plans, will also support the operable unit risk assessments and the risk assessments for the Comprehensive Site-Wide Operable Unit.

3.2 FATE AND TRANSPORT MODELING DATA

Fate and transport modeling data support the development and implementation of fate and transport models used at the FEMP to predict the migration of constituents from the site through environmental media. Fate and transport modeling is an integral part of the exposure assessment (Section 3.3). The types of data required for fate and transport modeling include information on the geology, hydrogeology, surface hydrology, and meteorology of the site and vicinity. These data are obtained from a variety of sources and are used in RI/FS risk assessments according to the following hierarchy:

- Data to be considered first: site-specific data obtained from the RI/FS database

- Data to be considered second: other site-specific data from sources such as the environmental monitoring annual reports, county soil surveys, and site-specific studies that complement the RI/FS characterization process (e.g., the Characterization Investigation Study)

- Data to be considered third: generic fate and transport modeling data from EPA reference documents. Examples of EPA reference documents that provide typical fate and transport modeling data include EPA 1988b, EPA 1989b, EPA 1987a, and EPA 1985a.

- Data to be considered fourth: generic fate and transport modeling data from secondary sources, subject to EPA approval

Sections 6.1 through 6.5 contain detailed presentations of the models, typical data values, and sources of data that are used in RI/FS risk assessments to predict the migration of constituents from the FEMP.

3.3 EXPOSURE ASSESSMENT DATA

Exposure assessment data are used to estimate gamma radiation exposures and intakes of chemicals and radionuclides by receptors. In addition to the results of fate and transport modeling, these data include values for parameters that quantitatively describe exposure scenarios such as ingestion rate, exposure frequency, exposure duration, biotransfer factors, absorption factors, averaging time, and body weight. Exposure assessment data are used in RI/FS risk assessments according to the following hierarchy:

- Data to be considered first: site-specific data obtained from the RI/FS database
- Data to be considered second: other regional and site-specific data from studies that complement the RI/FS characterization process
- Data to be considered third: generic exposure assessment data from EPA reference documents
- Data to be considered fourth: generic exposure assessment data from secondary sources, subject to EPA approval

Section 7.0 contains detailed presentations of the model equations, data values, and sources of data that are used for exposure assessments.

3.4 TOXICITY DATA

Toxicity data are used to quantify the human health hazard and hazard to ecological receptors from exposure to chemicals and radionuclides. The toxicity data used in RI/FS risk assessments are obtained from the following EPA sources:

- For carcinogens,
  - The EPA Health Effects Assessment Summary Tables (HEAST) for radionuclides (EPA 1991a)
  - The EPA Integrated Risk Information System (IRIS) for carcinogenic chemicals (EPA 1991b)
  - The EPA National Emission Standards for Hazardous Air Pollutants (NESHAPS) cancer risk coefficient per unit radiation dose (EPA 1989b)

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- For noncarcinogens,
  - The EPA IRIS database (EPA 1991b) and the most current HEAST data (EPA 1991a) for noncarcinogenic hazardous chemicals
  - Dose-response data from the open literature

If it is found that a reference dose is not available and toxicity data from the open literature must be used, estimated reference doses will be developed with the aid of EPA toxicologists. Section 8.0 contains specific references for the toxicity data used in RI/FS risk assessments.

3.5 UNCERTAINTIES

There are uncertainties associated with the information and data used in each phase of RI/FS risk assessments. These uncertainties are due to a number of factors, including parameter bias, parameter variability (random errors or natural variations), and improper model formulation. As EPA has pointed out in their guidance for health risk assessments, information is developed to determine what actions are necessary to reduce risks and not to eliminate all uncertainty from the analysis (EPA 1989a). Uncertainties associated with information and data will be evaluated in each risk assessment activity to provide the spectrum of information regarding the overall quality of the risk assessment. Additional discussions of uncertainties of the risk assessment process are given in Section 7.0 (exposure assessment), Section 8.0 (toxicity assessment), and Section 9.0 (risk characterization).

4.0

4.0 IDENTIFICATION OF CONTAMINANTS OF POTENTIAL CONCERN

4.1 ANALYSIS OF DATA

The analytical data obtained from the sources listed in Section 3.0 will be evaluated prior to use in the quantitative risk assessments. The criteria for evaluating the suitability of the data are based primarily on EPA guidance (EPA 1989a). These criteria are listed below:

- The methodology used to obtain concentration data and chemical forms will be considered. Data obtained via the following analytical methods are not considered appropriate for the quantitative risk assessment: (1) analytical methods that are not specific for a particular chemical or radionuclide (except total uranium), such as total organic carbon or total organic halogen, and (2) field screening instruments such as HNus, organic vapor analyzers, field instruments for detecting low energy radiation (FIDLERs), alpha-particle scintillation detectors, and Geiger-Mueller (GM) detectors. The methodology used to obtain specific data for the RI baseline risk assessment will be described in the RI reports.

- Sample quantitation limits associated with the analytical data will be identified if available. Unusually high sample quantitation limits will not be included in the data analysis if they cause the calculated exposure concentration to exceed the maximum detected concentration for a particular sample set.

- Matrix spike and matrix spike duplicate data will be analyzed in the RI/FS sampling data as stipulated in Volume 5 of the QAPP (DOE 1988a). Analytical results for chemicals will be reported using Contract Laboratory Program (CLP) data qualifiers. These qualifiers will guide the data's use in the quantitative risk assessment, as suggested in Exhibit 5-4 (EPA 1989a). Analytical results for radiological constituents will be reported as stipulated in the QAPP (DOE 1988a).

- Tentatively identified compounds (TICs) will be included in the analysis if historical site information suggests the TICs may have been present at the site, and when TICs appear often or TIC concentrations appear at high levels, further evaluation of TICs will be performed (EPA 1989a).

- Estimated quantitative results such as those identified by a "J" qualifier will be used in the risk assessment (EPA 1989a). The "J" qualifier is the most encountered data qualifier in Superfund data packages. Under the Contract Laboratory Program (CLP), the "J" Qualifier describes an estimated value either for a tentatively identified compound or when a compound is present (spectral identification criteria are met), but the value is less than the Contract Required Quantitation Limit (CRQL).

- If multiple dilutions are required to determine the value of a chemical present in high concentrations, and those dilutions result in unacceptable detection limits for other chemicals, only chemicals with positive detections (hits) will be considered from that analysis.

Of the data evaluated and found to be suitable for use in quantitative risk assessments, background concentration data are essential for identifying contaminants of potential concern. The use of background concentration data for this purpose is explained in the following sections.

4.2 STATISTICAL EVALUATION OF BACKGROUND DATA

Background concentration data are used to distinguish site-related contamination from naturally-occurring or other non-site-related levels of chemicals and radionuclides. Background concentration data obtained from the sources listed in Table 3-1 will be evaluated as part of the determination of contaminants of potential concern. The same background data will be utilized for all operable unit risk assessments as well as the site-wide risk assessments, until completion of the soil background sampling program, at which time the data acquired under that program will replace the regional soil background data.

Site-related concentration data for each constituent in each medium will be compared to the corresponding background concentration data. The comparison will be performed for each site-related concentration value as well as for the entire distribution of data for the specific constituent and medium.

At least twelve (12) background concentration values will be used for each constituent in each medium to determine the descriptive statistics of the background distribution, with at least 50% of the background data exceeding the sample quantitation limit (SQL). This number of samples meets the requirements of Ohio EPA's Closure Plan Review Guidance (OEPA 1990a) and exceeds the minimum number of samples recommended by Ohio EPA's "How Clean Is Clean" (OEPA 1991) policy on initial background sampling. This number also exceeds the number of samples recommended in EPA's Statistical Analysis of Ground Water Monitoring Data at RCRA Facilities (EPA 1989c).

4.2.1 Determination of Background Distribution

Each background data set will be evaluated to determine the probability distribution (normal, lognormal, or other) that best describes the data set. Two methods will be used to determine the distribution type.

In the first method, a histogram will be constructed from the data set and will be visually inspected to see if the distribution appears to be normal, lognormal, or other. Although this determination is subjective, the method complements inspection of data in tabular form or data that are summarized by descriptive statistics (such as the range, mean, median, and variance).

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Visual inspection of the histogram of the background data set is necessary when many of the data are non-detects.

The second method consists of the construction of a probability plot of the data set. If a straight line fits the plotted points reasonably well, a normal distribution will be assumed. If the data do not follow a straight line on the probability plot, the data will be log-transformed and replotted. If a straight line fits the log-transformed plot of the data, a lognormal distribution will be assumed. If a straight line does not fit the plotted points on either the normal probability plot or the log-transformed probability plot, then it will be assumed that the data set is neither normally distributed nor lognormally distributed. Although a visual inspection of the probability plot is often sufficient to determine whether the plotted points follow a straight line, a quantitative determination of the "linearity" of the data is performed.

The quantitative evaluation of the probability plots will be performed by calculating the correlation coefficient of the plotted points on the normal probability plot or on the lognormal probability plot. The correlation coefficient will be compared with a critical value that depends on sample size (n) and the chosen confidence level α (equal to 0.05) (Looney and Gulledge 1985). The values that the correlation coefficient must meet or exceed in order to conclude that the distribution is normal or lognormal are given in Table 4-1. The results of the two methods for assessing the type of distribution will determine the appropriate statistical treatment of background data for identifying contaminants of potential concern.

4.2.2 Treatment of Non-Detected Results for Background Concentrations

Analytical results are presented as "non-detects" whenever chemical concentrations in samples do not exceed the detection or quantitation levels for the analytical procedures for those samples. There are numerous terms used to describe the detection or quantitation levels (EPA 1989a). Sample quantitation limits (SQLs) are the most relevant quantitation limits for evaluating non-detected chemicals. SQLs take into account sample characteristics, sample preparation, and analytical adjustments. Generally, the detection limit (DL) (the lowest amount of a chemical that can be "seen" above the normal, random noise of an analytical instrument or method) is multiplied by a factor of three to five to obtain the SQL (EPA 1989a).

For radionuclides, the minimum detectable concentration (MDC) corresponds most directly to the SQL for chemicals. The MDC is the estimate of the activity concentration that can be practically achieved under a specified set of typical measurement parameters. These parameters include the sample size, counting time, counting efficiency, self-absorption and decay corrections, chemical yield, and other factors involved in determining activity concentrations (EPA 1980). For the

TABLE 4-1
CORRELATION COEFFICIENT TEST RESULTS AT A 95% CONFIDENCE LEVEL[a]

n	Value	n	Value
3	0.879	26	0.960
4	0.868	27	0.961
5	0.880	28	0.962
6	0.888	29	0.963
7	0.898	30	0.964
8	0.906	31	0.965
9	0.912	32	0.966
10	0.918	33	0.967
11	0.923	34	0.968
12	0.928	35	0.969
13	0.932	40	0.972
14	0.935	45	0.974
15	0.939	50	0.977
16	0.941	55	0.979
17	0.944	60	0.980
18	0.946	65	0.981
19	0.949	70	0.983
20	0.951	75	0.984
21	0.952	80	0.985
22	0.954	85	0.985
23	0.956	90	0.986
24	0.957	95	0.987
25	0.959	100	0.987

[a] (Looney and Gulledge 1985)

purposes of evaluating data in the RI/FS, the term "SQL" will be used for both chemicals and radionuclides.

Non-detected results (if present in the data set) must be considered with positively detected background results for determining the descriptive statistics for background data sets. Although EPA's Risk Assessment Guidance for Superfund Part A, Human Health Evaluation Manual allows for best professional judgement in determining the most appropriate assignment of values for non-detected results (EPA 1989a), EPA Region V has requested that a value of one-half the SQL be assigned for each non-detected result. Statistical treatment of background data for risk assessments will therefore conform with the methodology requested by EPA Region V.

A value of the SQL will be sought for each non-detected result. If SQLs cannot be obtained for chemical analytical results, the CRQL will be used as the value of the SQL. The uncertainty introduced by this assumption will be evaluated, since the CRQL may overestimate or underestimate the actual SQL (EPA 1989a).

4.2.3 Tests for Outliers in Background Concentration Data

An outlier is defined as an abnormally high or low data value. Since an outlier can represent a true extreme value or can indicate data errors, it is important to evaluate each data value to determine if it is an outlier or a true data value that will be included in the data set (Gilbert 1987).

Three methods will be used to evaluate data sets for the presence of outliers. In the first method, the histogram of the data set (see Section 4.2.1) will be visually inspected to see if any data points differ significantly from the remaining data. Usually a value that is four to five times as large as the remainder of the data is generally viewed with suspicion. A value that is an order of magnitude different from the other values can arise by the common error of misplacing a decimal (EPA 1989c). The second method consists of a visual inspection of the normal and lognormal probability plots of the data set (see Section 4.2.1). Any data points that differ significantly from the remaining data will be further evaluated.

The final method for identifying outliers in background concentration data sets is a quantitative test. Since this test, as with all quantitative tests for outliers, assumes a normal distribution, data that are not normally distributed will be transformed to approximate a normal distribution before the test is performed.

The final method to be used for identifying outliers consists of the following steps:

1. Calculate the mean, $\bar{x}$, and the standard deviation, s, of the data including all measurements.

2. Compute the statistic, T_n, given by

$$T_n = \frac{x_n - \bar{x}}{s} \qquad (4\text{-}1)$$

   for each value suspected of being on outlier.

3. Compare the statistic T_n to the critical value for the given sample size, n, from Table 4-2.

4. If the statistic T_n for the suspected value exceeds the critical value from Table 4-2, this is evidence that the suspected value, x_n, is a statistical outlier.

Since the presence of outliers can severely affect the determination of descriptive statistics and statistical comparisons, any potential or suspect outliers in background data sets will be investigated. The investigation will include, if possible, a review of the raw data associated with the determination of the background concentration value. Whenever possible, the background concentration for the suspect data point will be recalculated using the raw data and the appropriate calculation formula. Data transcription will also be checked for errors at each data entry step. When outliers cannot be attributed to errors, the descriptive statistics and statistical comparisons for the data set containing the outliers will be computed with and without the outliers to see if the two calculations are markedly different. Results that differ substantially due to the presence of outliers, will be presented both with and without outliers included.

4.3 SELECTION CRITERIA FOR CONSTITUENTS OF POTENTIAL CONCERN

Data available from the site investigation will be compared with background data to determine the constituents of potential concern. Since there is a large number of samples from various media that have analytical results for numerous chemicals and radionuclides, a systematic methodology will be implemented to compare site-related data to background data. Each site-related data value as well as the entire data set for a specific constituent in a specific medium will be compared to background data. Three methods of data comparison will be used. Any site-related data value or data set that cannot be determined to be due to background levels for the constituent in the specific medium will be further evaluated (Section 4.3.3). If further evaluation fails to demonstrate that the constituent is not site-related, then the constituent is considered to

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TABLE 4-2
CRITICAL VALUES FOR T_n (ONE-SIDED TEST)
(UPPER 5% SIGNIFICANCE LEVEL)[a]

Number of Observations	Critical Value	Number of Observations	Critical Value
3	1.153	30	2.745
4	1.463	35	2.811
5	1.672	40	2.866
6	1.822	45	2.914
7	1.938	50	2.956
8	2.032	55	2.992
9	2.110	60	3.025
10	2.176	65	3.055
11	2.234	70	3.082
12	2.285	75	3.107
13	2.331	80	3.130
14	2.371	85	3.151
15	2.409	90	3.171
16	2.443	95	3.189
17	2.475	100	3.207
18	2.504	105	3.224
19	2.532	110	3.239
20	2.557	115	3.254
21	2.580	120	3.267
22	2.603	125	3.281
23	2.624	130	3.294
24	2.644	135	3.306
25	2.663	140	3.318

[a] (ASTM 1991)

be a constituent of potential concern and an exposure assessment for the constituent will be performed. The tests to identify outliers described in Section 4.2.3 will be performed for site-related data and outliers will be investigated.

4.3.1 Comparison of Individual Data Values to Background

The first test to determine if a constituent is site-related will be to compare each data value for a constituent and medium to an upper tolerance limit (UTL) calculated from the background data for that constituent in the same medium. The method for constructing the UTL is taken from EPA guidance, Statistical Analysis of Ground-Water Monitoring Data at RCRA Facilities (1989c). The UTL will be calculated by one of two methods, depending on whether the background distribution is normal or lognormal. (This test will not be performed for background data distributions that are neither normal nor lognormal.)

For normal distributions of background data, the UTL will correspond to the value of the upper 95% confidence limit on the 95th quantile of the background distribution and will be calculated as (EPA 1989c).

$$UTL = \bar{x} + (K)(s) \qquad (4\text{-}2)$$

where

$\bar{x}$ = arithmetic mean of the background samples
K = tolerance factor for estimating the upper 95% confidence limit on the 95th quantile of a normal distribution, from Table 4-3
s = sample standard deviation.

For lognormal distributions of background data, the UTL will be calculated as (Gilbert 1987):

$$UTL = e^{(\bar{y} + zs_y)} \qquad (4\text{-}3)$$

where

$$\bar{y} = \frac{1}{n}\sum \ln x \qquad (4\text{-}4)$$

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TABLE 4-3
TOLERANCE FACTORS (K) FOR ONE-SIDED NORMAL TOLERANCE INTERVALS FOR 95% CONFIDENCE LIMIT ON THE 95TH QUANTILE

n	K	n	K
3	7.656	22	2.350
4	5.144	23	2.329
5	4.210	24	2.309
6	3.711	25	2.292
7	3.401	30	2.220
8	3.188	35	2.166
9	3.032	40	2.126
10	2.911	45	2.092
11	2.815	50	2.065
12	2.736	60	2.022
13	2.670	70	1.990
14	2.614	80	1.965
15	2.566	90	1.944
16	2.523	100	1.927
17	2.486	120	1.899
18	2.453	145	1.874
19	2.423	300	1.800
20	2.396	500	1.763
21	2.371		

[a] (Owen 1962)

such that $e^{(\bar{y})}$ is the geometric mean, and

z = 1.645 (95% confidence limit for one-tailed test) (Pearson and Hartley 1966)

$$s_y = \sqrt{\frac{\sum (y-\bar{y})^2}{n-1}} \qquad (4-5)$$

such that $e^{(S_y)}$ is the geometric standard deviation.

Each data value will be compared to the appropriate UTL for the constituent and medium. Any data value which exceeds the UTL indicates that the constituent may be a contaminant of potential concern for that medium and will be evaluated according to the criteria of Section 4.3.3. If all data values for a constituent and medium are less than the UTL, or if the background data distributions are neither normal nor lognormal, then other methods that compare the data values (as a data set) for the constituent and medium with the background data set will be used. These methods are described in the next section.

4.3.2 <u>Comparison of Data Sets to Background Data Sets</u>

As noted in the preceding section, if each data value from a data set does not exceed the UTL, or if the UTL cannot be constructed for the background data (if background data distributions are neither normal nor lognormal, or if a large percentage of the background data set are non-detects), two additional tests will be made on the data set for a specific constituent and medium. If either of the two tests is "failed" by the data set, then the specific constituent may be a contaminant of potential concern and will be evaluated according to the criteria of Section 4.3.3. If both tests are passed by the data set, then the specific constituent is not considered further (since the individual values from the data set have also passed the comparison test described in Section 4.3.1).

The two tests that will be performed are the Wilcoxon Rank Sum (WRS) test and the Quantile test (EPA 1990l). Both of these are nonparametric tests that do not require the background distribution and the site distribution to be normal or lognormal. Each test is used to assess whether the site data distribution differs from the background data distribution.

Wilcoxon Rank Sum Test

The WRS test consists of ordering (ranking) the combined background data and site data, finding the sum of the ranks of the site data, and computing a test statistic. If that statistic is sufficiently large, then the constituent may be a contaminant of potential concern and will be evaluated according to the criteria of Section 4.3.3. The WRS test can be used even when there is a moderately large number of site data values reported as non-detects. The following is a brief description of the WRS test. A detailed explanation of the test is given in Statistical Methods for Evaluating the Attainment of Cleanup Standards, Volume 3: Background-Based Standards for Soils and Solid Media (EPA 1990b).

The null and alternative hypotheses related to the WRS test are as follows:

H_0 (null hypothesis): $Pr = ½$

H_a (alternative hypothesis): $Pr > ½$

where

Pr = Probability that a concentration measurement of a sample collected at a random location at the site is greater than a concentration measurement of a sample collected at a random location in the background area.

H_0 is assumed to be true unless the test indicates that H_0 should be rejected in favor of H_a. When H_0 is true, the distribution of concentration measurements in the background area is the same shape and location as the distribution at the site, indicating that the site is not contaminated with the given constituent.

The steps that will be followed for the WRS test are:

1. Specify the value of α (Type I error rate) as equal to 0.05.
2. Combine the values for the "m" samples from the background area and the values for the "n" samples from the site into one data set.
3. Consider all data ($N = m+n$) as one data set and rank the N data from 1 to N from the lowest to the highest concentration.
4. If data are tied (i.e., have the same value) assign them the midrank, that is the average of the ranks that would otherwise be assigned to those data.
5. Non-detects are assigned a rank less than the rank of the smallest measured value in the combined data set.
6. Sum the ranks of the n site data.
7. Compute the test statistic for the rank sum using the appropriate formula (EPA 1990b).

8. Compare the test statistic to the cumulative normal distribution statistic, z, for α = 0.05 (i.e., z = 1.645). If the test statistic for the rank sum exceeds 1.645, then we will conclude that the constituent in that medium may be a contaminant of potential concern and will be evaluated according to the criteria of Section 4.3.3. If the statistic for the rank sum does not exceed 1.645, then we will perform the Quantile test of the data.

Quantile Test

The Quantile test is initiated by ordering the combined background and site data as done for the WRS test. A count is made of the number of measurements from the site that are in the largest 100 (1-q)% of the combined set of measurements, where "q" depends on the sample sizes. A test statistic is computed, to which the number of measurements from the site in the largest 100 (1-q)% of the combined set of measurements is compared. If the test statistic is exceeded, then the constituent may be a contaminant of potential concern and will be evaluated according to the criteria of Section 4.3.3. If the test statistic is not exceeded, then the constituent is not considered to be a contaminant of potential concern. The Quantile test will be conducted in accordance with the guidance given in Statistical Methods for Evaluating the Attainment of Cleanup Standards, Volume 3: Background-Based Standards for Soils and Solid Media (EPA 1990b).

4.3.3 Other Criteria for Selecting Constituents of Potential Concern

Constituents that are determined to require further evaluation, as an outcome of the tests performed according to the methodology of Section 4.3.1 and Section 4.3.2, will be excluded as chemicals of potential concern if any one of the following criteria are met. Conditions for these specific exclusions are given in EPA guidance (EPA 1989a).

Chemicals that are: (1) essential human nutrients such as sodium, potassium, magnesium, calcium, and iron, (2) present at low concentrations (i.e., only slightly above naturally-occurring levels), and (3) toxic only at very high doses (i.e., much higher than those that could be associated with the site) will not be identified as chemicals of potential concern (EPA 1989a). Concentrations of essential nutrients in each operable unit will be compared to background concentrations according to the UTL and the non-parametric tests described in Section 4.3.2 in order to determine constituents of potential concern with respect to items (2) and (3) listed above. This elimination criterion will not be applied to radioactive isotopes of the essential nutrients.

Chemical constituents will not be identified as a chemical of potential concern if it is a common laboratory contaminant and if all sample concentration results are less than ten (10) times the highest blank concentration. Common laboratory contaminants include acetone, 2-butanone,

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methylene chloride, toluene, and the phthalate esters. Other chemicals will be eliminated if all results are less than five times the highest concentration detected in a blank. Chemicals considered common laboratory contaminants, which may be actual constituents of potential concern at the site, will be considered on a case-by-case basis.

Whenever there is a large number of constituents that are tentatively identified as chemicals of potential concern, a concentration-toxicity screening procedure (EPA 1989a) will be used to identify constituents in a particular medium that are most likely to contribute significantly to risks calculated for exposure scenarios involving that medium. This procedure will not be used for radionuclides at the FEMP. In the concentration-toxicity screening procedure, a risk factor is calculated by multiplying the maximum detected concentration of the constituent by its toxicity value, i.e., either the slope factor or the inverse of the reference dose (1/RfD). In other words, the screening is performed using the following:

$$R_{ij} = (C_{ij})(T_i) \qquad (4\text{-}6)$$

where

R_{ij} = risk factor for the ith chemical in the jth medium
C_{ij} = maximum detected concentration of the ith chemical in the jth medium
T_i = toxicity value for the ith chemical (1/RfD for noncarcinogens or the cancer slope factor for carcinogens)

From these values the total risk factor for a medium, R_j, is calculated as

$$R_j = \sum_i R_{ij} = \sum_i (C_{ij})\,(T_i) \qquad (4\text{-}7)$$

Separate total risk factors are calculated for carcinogenic and noncarcinogenic effects for each chemical. The ratio of the chemical-specific risk factor (R_{ij}) to the total risk factor (R_j) approximates the relative contribution to the overall risk for each constituent in the medium. Chemicals for which

$$\frac{R_{ij}}{R_j} < 0.01$$

will be eliminated from further consideration in the quantitative risk assessment (EPA 1989a). Application of this toxicity-screening procedure for each operable unit or site-wide risk assessment, will be subject to EPA approval on a case-by-case basis.

All chemicals identified as chemicals of potential concern prior to screening for human health risk will be evaluated in the ecological assessment. Because ecological receptors currently have access to the FEMP site, no distinction will be made between present and future chemicals of potential concern, as will be the case in the human health risk assessment.

4.4 CHEMICALS AND RADIONUCLIDES AT THE FEMP

Constituents detected or inferred thus far in the RI/FS process are listed in Table 4-4. Many, but not all, short-lived radioactive progeny of long-lived radionuclides are assumed to be present and are listed in the table. These tabulations are based on work that has been performed to date on RI/FS risk assessments and are not all inclusive. Analytical results from ongoing site characterization studies may lead to a revision of Table 4-4. This is particularly true for Operable Units 3 and 5, which have been redefined to include areas and facilities outside of the original scope of the FEMP RI/FS.

TABLE 4-4
RADIONUCLIDES AND HAZARDOUS CHEMICALS IN ENVIRONMENTAL MEDIA OR OPERABLE UNIT SOURCE TERMS

X = Detected or inferred
-- = Not detected or inferred

Analytes	Operable Unit 1	Operable Unit 2	Operable Unit 3[a]	Operable Unit 4	Operable Unit 5
Radionuclides					
Ac-227	--	--	--	X	--
Cs-137	X	X	X	--	X
Np-237	X	--	--	--	--
Pa-231	--	--	--	X	--
Pb-210	X	X	--	X	--
Pu-238	X	X	X	--	X
Pu-239/240	X	--	--	--	--
Ra-224	--	--	--	X	--
Ra-226	X	X	X	X	X
Ra-228	X	X	X	X	X
Rn-220	X	X	X	X	X
Rn-222	X	X	X	X	X
Sr-90	X	X	X	--	X
Tc-99	X	--	X	--	X
Th-228	X	X	X	X	X
Th-230	X	X	X	X	X
Th-232	X	X	X	X	X
U-234	X	X	X	X	X
U-235/236	X	X	X	X	X
U-238	X	X	X	X	X
Inorganics					
Aluminum	X	X	--	X	X
Arsenic	X	X	--	X	X
Antimony	--	--	--	--	X
Barium	X	X	--	X	X

TABLE 4-4
(Continued)

X = Detected or inferred
-- = Not detected or inferred

Analytes	Operable Unit 1	Operable Unit 2	Operable Unit 3[a]	Operable Unit 4	Operable Unit 5
Beryllium	X	X	--	X	X
Cadmium	X	X	X	X	X
Calcium	--	--	--	X	--
Chromium	X	X	--	X	X
Cobalt	X	--	--	X	X
Copper	X	X	--	X	X
Iron	--	--	--	X	X
Lead	X	X	X	X	X
Magnesium	X	--	--	X	X
Manganese	X	X	--	X	X
Mercury	X	X	--	X	X
Molybdenum	X	X	--	--	--
Nickel	X	X	--	X	X
Potassium	--	--	--	X	--
Selenium	X	X	--	X	X
Silver	X	X	X	X	X
Sodium	--	--	--	X	--
Thallium	X	--	--	X	--
Vanadium	X	X	--	X	X
Zinc	X	X	--	X	X
Organics					
1,1-dichloroethane	X	X	X	--	X
1,1-dichloroethene	--	--	X	--	X
1,1,1-trichloroethane	X	X	X	--	X
1,1,2-Trichloro-1,2,2-trifluoroethane	--	X	--	--	--
1,1,2,2-Tetrachloroethane	--	--	X	--	X
1,2-dichloroethene	--	--	X	--	X

TABLE 4-4
(Continued)

X = Detected or inferred
-- = Not detected or inferred

Analytes	Operable Unit 1	Operable Unit 2	Operable Unit 3[a]	Operable Unit 4	Operable Unit 5	
2-Butanone	X	X	X	--	X	1
2-Methylnapthalene	X	X	X	--	X	2
2-methylphenol	--	X	--	--	--	3
2-propanol	X	--	--	--	--	4
2,4-dimethylphenol	--	X	--	--	--	5
4-methyl-2-pentanone	--	--	X	--	X	6
4-methylphenol	--	X	--	--	--	7
Acenaphthene	X	X	X	--	X	8
Acetone	X	X	--	--	--	9
Anthracene	X	X	X	--	X	10
Benzene	--	X	X	--	X	11
Benzo(a)anthracene	X	X	X	--	X	12
Benzo(a)pyrene	X	X	X	--	X	13
Benzo(b)fluoranthene	X	--	X	--	X	14
Benzo(g,h,i)perylene	X	X	X	--	X	15
Benzo(k)fluoranthene	X	X	X	--	X	16
Benzoic acid	--	--	X	--	X	17
Beta-BHC	--	--	X	--	X	18
Bis(2-ethylhexyl) phthalate	X	X	X	--	X	19
Butyl benzyl phthalate	X	X	--	--	--	20
Carbon disulfide	X	X	X	--	X	21
Chlordane	--	X	--	--	--	22
Chlorobenzene	--	--	X	--	X	23
Chloroform	X	X	X	--	X	24
Chrysene	X	X	X	--	X	25
cis-1,2-dichloroethene	X	--	--	--	--	26
Cyanide	--	--	X	X	--	27

TABLE 4-4
(Continued)

X = Detected or inferred
-- = Not detected or inferred

Analytes	Operable Unit 1	Operable Unit 2	Operable Unit 3[a]	Operable Unit 4	Operable Unit 5
DDT	X	--	--	--	--
Di-n-butyl phthalate	X	X	X	--	X
Di-n-octyl phthalate	X	X	--	--	--
Dibenzo(a,h)anthracene	--	X	X	--	X
Dibenzofuran	X	X	X	--	X
Ethyl parathion	X	--	--	--	--
Ethyl benzene	X	X	X	--	X
Fluoranthene	X	X	X	--	X
Fluorene	X	X	X	--	X
Indeno(1,2,3-cd)pyrene	X	X	X	--	X
Methyl parathion	X	--	--	--	--
Methylene chloride	X	X	--	--	--
N-nitrosodiphenylamine	--	X	X	--	X
Naphthalene	X	X	X	--	X
PCBs (Aroclors-1242, 1248, 1254, 1260)	X	X	X	--	X
Pentachlorophenol	X	--	X	--	X
Phenanthrene	X	X	X	--	X
Phenol	X	X	X	--	X
Pyrene	X	X	X	--	X
Tetrachloroethane	X	--	--	--	--
Tetrachloroethene	X	X	X	--	X
Toluene	X	X	--	--	--
Total Xylenes	X	X	X	--	X
Trichloroethene	X	--	X	--	X
Vinyl chloride	--	--	X	--	X

[a] Operable Unit 3 is presently insufficiently characterized. The contaminants present in the soil, perched water, and groundwater beneath the production area are assumed to be present in the buildings as well.

5.0

5.0 DEVELOPMENT OF EXPOSURE SCENARIOS

This section defines and describes the components of an exposure scenario, discusses the steps involved in identifying and developing exposure scenarios, and proceeds through screening and selection of currently identified exposure scenarios for the FEMP. Selected exposure scenarios are those that are determined to require a quantitative evaluation in the risk assessment.

Components of an exposure scenario include a source of contaminants, mechanisms that facilitate the transport of contaminants from sources through various media, receptors in the local environment, and a route or mechanism for exposure of those receptors.

Steps involved in developing exposure scenarios include characterization of the exposure setting, identification of potential exposure pathways, and selection of site-specific exposure pathways to be quantitatively evaluated in the risk assessment. Section 5.1 addresses the character of the site setting within which potential exposures could occur. Section 5.2 discusses potential environmental transport and exposure mechanisms at the site. Section 5.3 discusses the methodology for selecting those pathways that will be quantitatively evaluated in the risk assessment. Section 5.4 discusses the receptors at or near the FEMP.

5.1 CHARACTERIZATION OF EXPOSURE SETTING

The first step in developing exposure scenarios is evaluating the site setting in which potential exposures could occur. The site setting is evaluated first in the development of exposure scenarios because characteristics of the site setting influence the types of transport mechanisms that could occur at the site and the types of receptor exposures that could occur in the vicinity of the site. Evaluation of the site setting involves examining the physical environment of the site and populations in the vicinity (receptors) that could be subject to potential exposures.

5.1.1 Physical Environment

A detailed description of the physical environment will be presented in the RI reports for the FEMP and addresses aspects of the local geography, surface topography, demographics, geology and hydrogeology, and ecology. A summary description of the physical environment at the FEMP is given in this section.

5.1.1.1 Geography

The FEMP is located on 1050 acres of land in rural areas of Hamilton and Butler counties in southwestern Ohio. The facility is located approximately 20 miles northwest of Cincinnati, Ohio.

The villages of Fernald, New Baltimore, Ross, New Haven, and Shandon are located within a few miles of the FEMP.

5.1.1.2 <u>Surface Topography</u>

The main physiographic features in the area are gently rolling uplands, steep hillsides along the major streams, and the Great Miami River Valley, which is a relatively broad, flat-bottomed valley flanked on either side by bluffs that rise to a maximum of 300 feet above the general level of the valley floor. Maximum elevation along the northern boundary of the FEMP property is a little more than 700 feet above mean sea level (MSL). The production area and waste storage area rest on a relatively level plain at about 580 feet MSL. The plain slopes from 600 feet MSL along the eastern boundary of the FEMP to 570 feet MSL at the K-65 silos, and then drops off toward Paddys Run at an elevation of 550 feet MSL. Drainage on the FEMP is generally from east to west into Paddys Run. One exception is the extreme northeast corner of the FEMP which drains east toward the Great Miami River.

5.1.1.3 <u>Surface Hydrology</u>

The primary surface drainage feature of the FEMP is Paddys Run, an intermittent stream. A tributary of the Great Miami River, Paddys Run, flows from north to south near the western boundary of the FEMP property (Figure 5-1). Paddys Run has historically received direct runoff from the western areas of the FEMP, including the silos and waste storage areas. One branch of Paddys Run, now known as the Storm Sewer Outfall Ditch, drains the southern end of the production area and feeds into Paddys Run approximately 650 feet upstream of the southern boundary of the FEMP.

5.1.1.4 <u>Demographics</u>

As an inactive industrial property undergoing characterization, remediation, and closure, there are no residences on the FEMP. The on-property worker population includes employees of DOE, WEMCO and other contractors. Workers are generally on the FEMP approximately eight hours per day, five days per week. Structures housing on-property workers are on approximately 300 acres in the center of the FEMP in the administration area and the production area.

Scattered residences and several villages, including Fernald, New Baltimore, Ross, New Haven, and Shandon, are located near the FEMP. Downtown Cincinnati is approximately 20 miles southeast of the FEMP and the cities of Hamilton and Fairfield are six to eight miles to the northeast. There is an estimated population of more than 24,000 within five miles of the center of the FEMP. The nearest resident is within three quarters of a mile (1200 meters) from the center of the facility. The nearest residences to the western FEMP property boundary (the

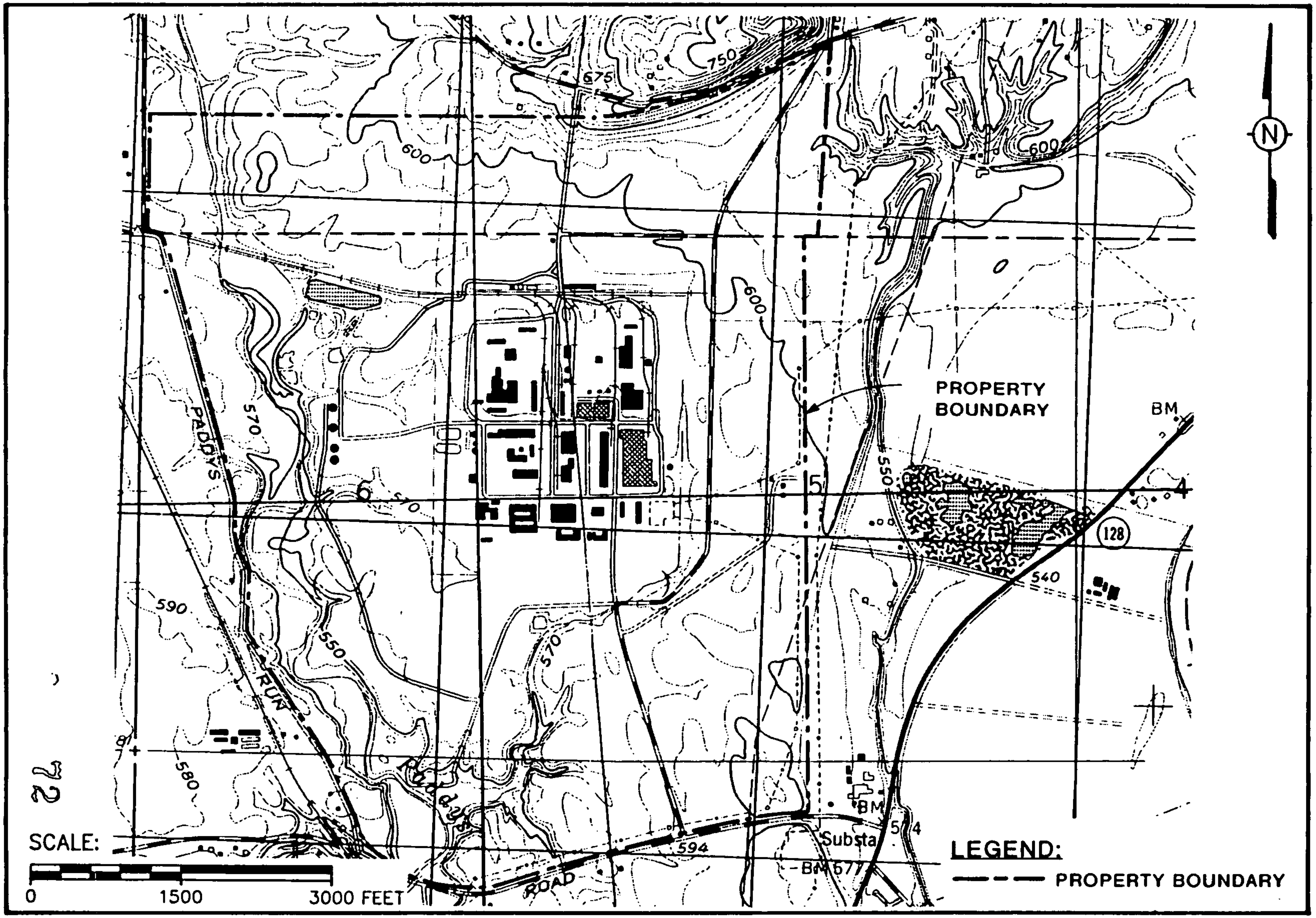

FIGURE 5-1 TOPOGRAPHIC MAP OF THE FEMP

boundary along the eastern side of Paddys Run Road) are located along the western side of Paddys Run Road. The Knollman Dairy Farm is located on Willey Road just outside the southeast corner of the FEMP property boundary (leased grazing areas include areas inside the property boundary). Several residences are located off Paddys Run Road approximately one-half mile south of the FEMP property boundary and along New Haven Road approximately one mile south of the FEMP property boundary. These residences are in the vicinity of the South Plume, a portion of the Great Miami Aquifer that contains a plume of uranium contamination which extends south of the FEMP property boundary approximately three-quarters of a mile.

5.1.1.5 Historical Significance

The area surrounding the FEMP contains several sites of historical interest. The National Register of Historic Places lists five prehistoric Indian sites within three miles of the FEMP. These include the Adena Circle, the Hogen-Borger Mound, the Demoret Mound, the Colerain Work, and the Dunlap Work. The State Historical Preservation Officer reports that there are no known sites of archaeological significance on the FEMP.

5.1.1.6 Geology and Hydrogeology

The FEMP site is located on a dissected till plain left by Wisconsin Glaciation. This plain overlays a two- to three-mile-wide subterranean valley known as the New Haven Trough. This valley formed as a result of Pleistocene glaciation and subsequently filled with glacial outwash materials and till. The buried valley is approximately one-half to more than two miles wide and is U-shaped, having a broad, relatively flat bottom and steep valley walls. Interbedded glacial overburden deposits occur within the outwash deposits, but in most cases are of limited lateral extent. The overburden deposits are composed primarily of poorly sorted pebbles, cobbles, and boulders in a predominantly clay matrix.

Within the glacial overburden deposits there are numerous perched water-bearing zones that have limited interconnection. The majority of these perched zones are of glaciofluvial origin and consist of small beds of highly sorted sands and gravels. These beds are probably the result of small meltwater streams that occurred along the ice margin and within the glacier itself. These intertill aquifers have the following general characteristics:

- High variability in areal extent, thickness, and volume
- Based upon hydrograph analysis, limited interconnection between the intertill aquifers

- The majority are confined by layers of relatively impermeable till. This results in conditions where water will rise in a well to a level higher than where the water was first encountered (confined or artesian conditions).

- Hydraulic conductivities are highly variable with an expected range of 2.8 x 10^{-2} to 280 ft/day (10^{-5} to 0.1 cm/s) (Freeze and Cherry 1979). At the FEMP, series of slug tests of water-bearing zones in the till found hydraulic conductivities ranging from 1.6 ft/day (5.6 x 10^{-4} cm/s) in Well 1048 to 7.1 x 10^{-3} ft/day (2.5 x 10^{-6} cm/s) in Well 1079.

- Porosities range from 22.1 percent to 36.7 percent, with a mean of 31 percent (Morris and Johnson 1967).

Generally these glaciofluvial interbeds are considered to be the major water-bearing units within the glacial overburden. However, movement of water and contaminants within these units is constrained because of the limited extent and interconnection of these units.

The Great Miami River has eroded through the glacial overburden and is now in direct contact with the glaciofluvial outwash deposits that comprise the buried valley aquifer. Paddys Run is also in contact with these deposits in its lower reaches. Within some areas, overburden deposits overlie the bedrock uplands and portions of the outwash materials where they form the thick unconsolidated sediment layers beneath the soil zone. This glacial overburden is composed of dense, silty clay that varies in composition vertically and laterally. The silty clay overburden contains lenses of poorly sorted fine- to medium-grained sand and gravel, silty sand, and silt with layers of silty clay.

The bedrock in the vicinity of the FEMP consists of predominantly flat-lying, olive-gray Ordovician shales with thin, interbedded layers of limestone. This shale forms the buried valley walls of the New Haven Trough. The buried valley is generally carved into this shale between 60 and more than 200 feet below the pre-erosional land surface in the vicinity of the FEMP.

Three flow systems of the Great Miami Aquifer converge in the vicinity of the FEMP reservation. As shown in Figure 5-2, groundwater in the Dry Fork Section of the New Haven Trough generally flows from west to east. Groundwater in the Shandon Tributary of the New Haven Trough generally flows to the southeast, and groundwater in the Ross Section of the New Haven Trough generally flows to the southwest. Figure 5-2 also shows a flow divide located in the southern portion of the FEMP that separates Dry Fork Section groundwater from Shandon Tributary groundwater. The location of the divide fluctuates, depending on flow conditions; therefore mixing occurs along the divide.

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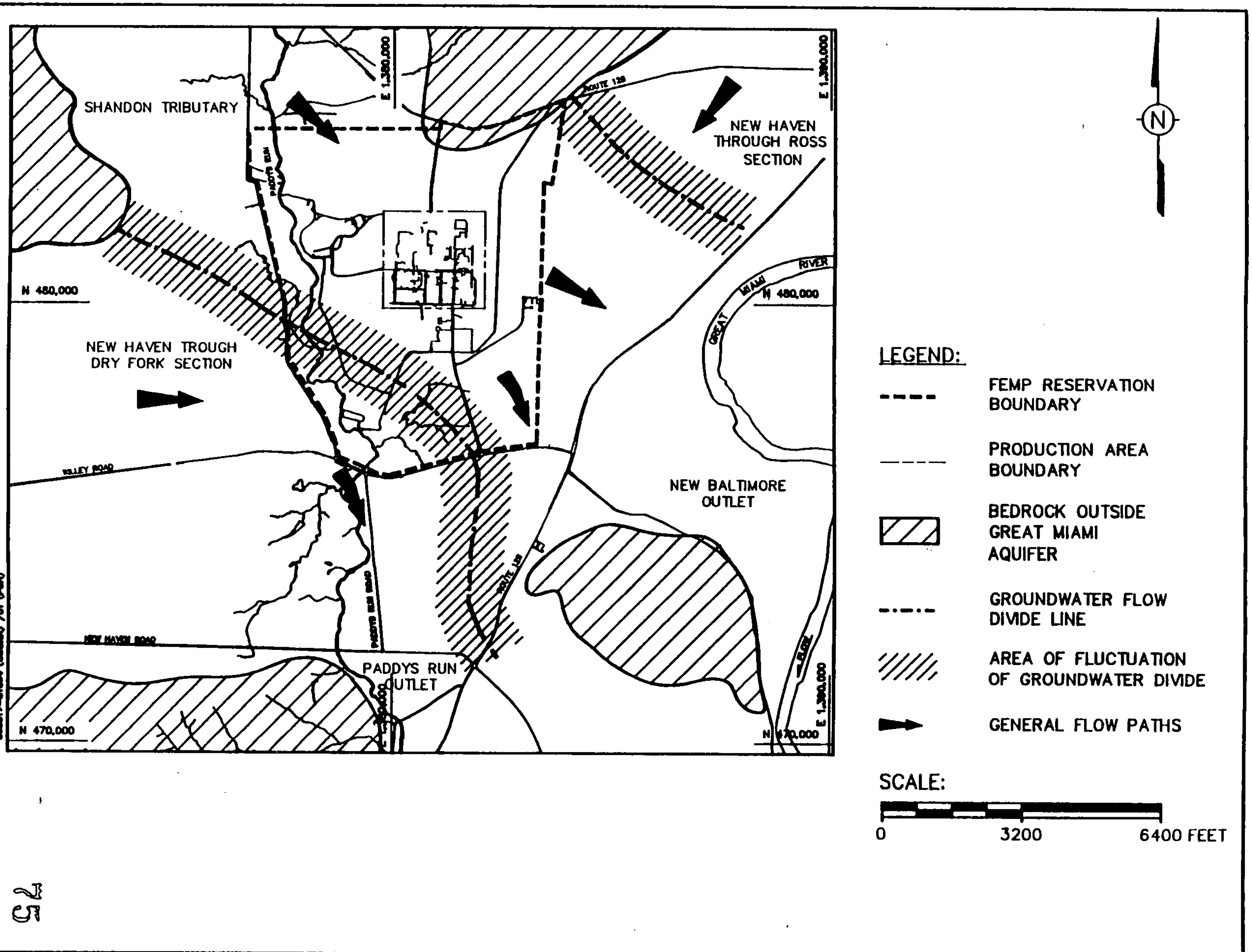

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FIGURE 5–2. GROUNDWATER FLOW DIVIDES

Groundwater from the Ross Section does not pass beneath the FEMP. A flow divide separating the Ross Section groundwater from Shandon Tributary groundwater is located east of the FEMP, as shown in Figure 5-2. This divide is influenced by pumping of the collector wells located within and near the "big bend" of the Great Miami River.

The surface and subsurface hydrology of the site are directly connected at various locations. Paddys Run loses flow to the top of the regional aquifer, which intersects the stream bed within the site boundaries. Natural gradients cause the groundwater beneath the FEMP to exit the study area by either flowing east to the Great Miami River (upstream from New Baltimore), or by flowing south through the branch of the bedrock channel west of New Baltimore. In either case, the Great Miami River is the ultimate receptor of groundwater from the study area.

Groundwater is the source of water for industrial and domestic use in the area. The estimated pumping from the major well fields in the area averages approximately 18 million gallons per day (mgd). Additionally, there are smaller industrial, commercial, agricultural, and private groundwater users in the area.

The residences in the area use either domestic wells or cisterns for water supplies. Generally, cisterns are used in areas underlain by bedrock. Many residents use bottled water for drinking because of the bad taste and smell of the water from some parts of the aquifer. Wells downgradient from the FEMP are generally completed in the upper part of the aquifer and pump only when there is a demand for water for domestic washing and sanitation.

There are several large farms in the vicinity of the FEMP that use groundwater. Two known irrigation wells on farms east of the site and northwest of Route 128 are currently being used for field irrigation. One farm on New Haven Road south of the property, between Route 128 and the village of New Baltimore, also is known to irrigate from a well on the property. Those farmers east and south of the FEMP, who are in close proximity to the Great Miami River, irrigate their fields with water from the river (Plummer 1990).

5.1.1.7 <u>Ecological Setting</u>

This section describes the major habitats at and adjacent to the FEMP. Ecological receptors are described in detail in Section 5.1.2.3.

The FEMP lies in the Oak-Hickory Forest Section of the Eastern Deciduous Forest, as described by Bailey (1978). Ecological communities at the FEMP have been described by Facemire et al. (1990) as consisting of grazed and ungrazed pastures, two pine plantations, deciduous woodlands, riparian woodlands, and a "reclaimed fly ash pile area," referred to in RI/FS documents as the Inactive Fly Ash Disposal Area (Figure 5-3). Forested jurisdictional wetlands, as defined by federal guidance (FICWD 1989), were delineated as part of the RI/FS and occupy approximately 50 acres north of the production area (Figure 5-4). Emergent jurisdictional wetlands, also included in the RI/FS study, occur along the railroad spur and various drainageways on the FEMP. Paddys Run and adjacent aquatic habitats harbor small fish, amphibians, and a variety of benthic macroinvertebrates. The most common fish are the bluntnose minnow, creek chub, and stoneroller minnow (Facemire et al. 1990). The most common benthic macroinvertebrates are non-biting midges, riffle beetles, mayflies, and stoneflies.

A total of 47 species of trees and shrubs, 190 species of herbaceous plants, 20 mammal species, 98 bird species, 10 species of amphibians and reptiles, 21 species of fish, 47 families of benthic macroinvertebrates, and 132 families of terrestrial invertebrates were found at the FEMP by Facemire et al. (1990).

Organisms in the Great Miami River adjacent to the FEMP have been characterized by Ohio Environmental Protection Agency (OEPA) (1982a, 1989), Miller et al. (1987, 1988, 1989), and by the U.S. Geological Survey (USGS 1974 to 1982). A total of 106 species of fish has been recorded from the Great Miami River from 1900 to 1978 (Trautman 1957, 1981), while OEPA collected 76 species in their most recent survey of the river (OEPA 1989). No federally listed threatened or endangered species have been observed on the FEMP or in its immediate vicinity. Suitable habitat for one species of mammal listed as federally endangered, the Indiana bat, was located along Paddys Run during RI/FS studies, but the species was not found on site. The range of the cave salamander, a state endangered species, overlaps the FEMP, but was not found during RI/FS studies.

5.1.2 Potential Sources of Contaminants at the FEMP

The FEMP is a large inactive industrial facility containing both radioactive and hazardous wastes (Section 4.4). Principal radioactive constituents include, but are not limited to, unknown quantities of thorium-232 and uranium-238 and their associated progeny. The equilibrium of these decay chains has generally been disturbed due to removal of some progeny during processing operations. Principal hazardous waste constituents include heavy metals, chlorinated and nonchlorinated solvents, polychlorinated biphenyls (PCBs), and polycyclic aromatic hydrocarbons. The source areas for nonradioactive constituents are often of smaller areal extent than the radioactive constituents. The bulk of the process wastes were disposed in either the waste pits or the silos on property (Section 2.3). There are a multitude of contamination sources on property including open waste pits (containing contaminated wastes and water), contaminated

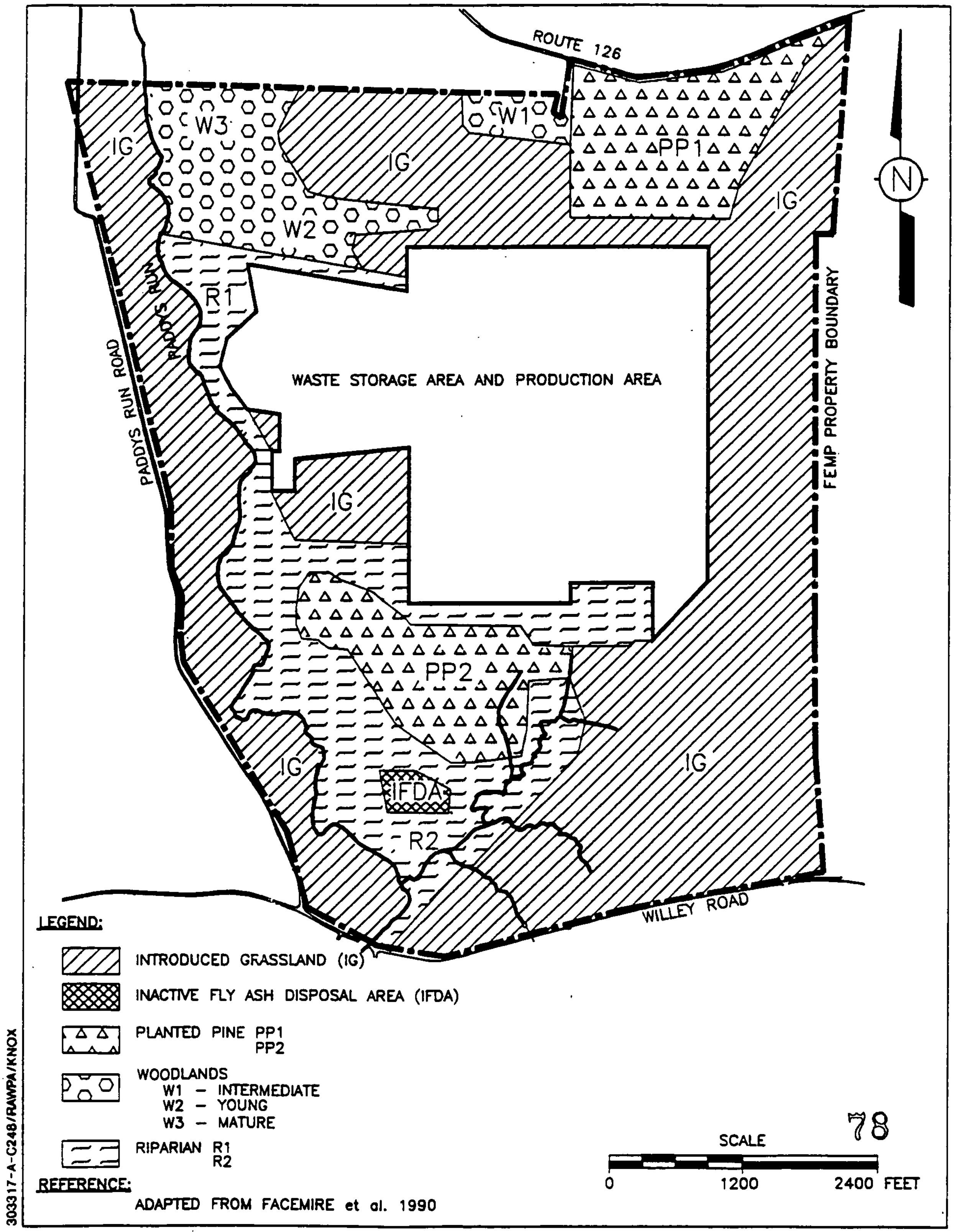

FIGURE 5-3. HABITAT TYPES PRESENT ON THE FEMP

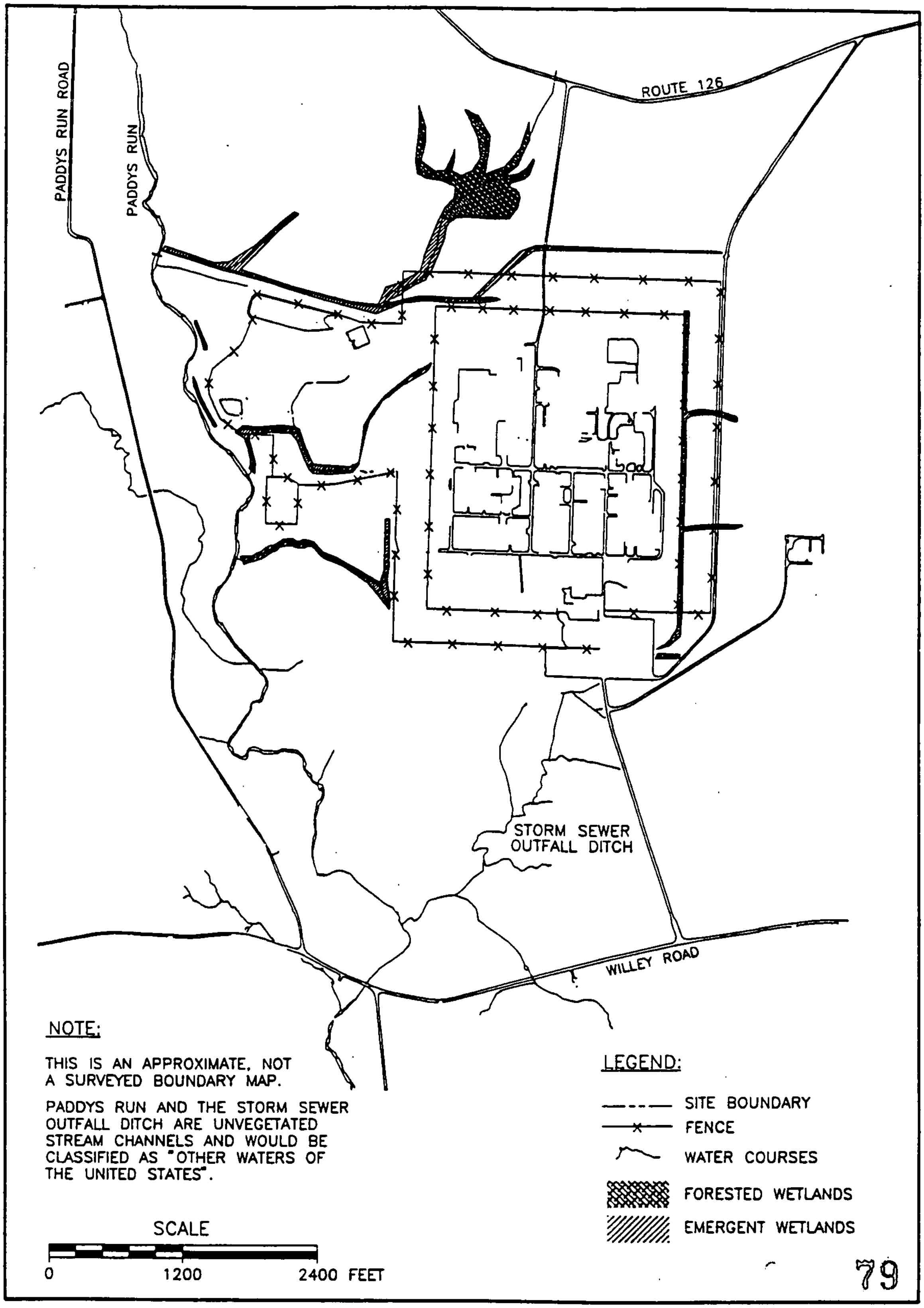

FIGURE 5-4. APPROXIMATE BOUNDARIES OF JURISDICTIONAL WETLANDS AT THE FEMP

soils, buried wastes, and contaminated buildings. Potential sources of contaminants at the FEMP are presented in Table 5-1. These sources are consistent with the revised operable unit definitions presented in Section 1.7 of this addendum. Radioactive decay and environmental degradation of contaminants within these source areas will be considered in the risk assessments.

5.1.3 Land Use

The land within the FEMP property boundaries currently contains a large, inactive industrial facility. Many of the facility's buildings are currently used for storage of idle process equipment. Administration and laboratory operations conducted at the site are currently focused on the safe shutdown of the facility and the environmental restoration of the property. A security fence surrounds the entire FEMP property, and a second line of fences surrounds several internal areas, including the production area and the waste disposal area. These fences are regularly patrolled by a large, full-time security force. These active (security patrols) and passive (fences) access restrictions are currently in place at the FEMP. Over the past 40 years, these controls have proven to be effective for restricting unauthorized site access to transient forays of limited duration (intruders). No hunting or fishing is allowed on the site, but approximately 400 acres of the site are leased to a nearby resident for grazing of cattle.

Land use surrounding the FEMP is mainly agricultural, with dairy, beef, corn, and soy bean production. Several industries, including Delta Steel, Albright & Wilson Chemical Company, Ruetgers-Nease Chemical Company, two commercial gravel operations, and a cement plant, are located to the south. The Miami Whitewater Forest and a Hamilton County park are located within five miles of the FEMP.

5.1.4 Potentially Exposed Populations

Determination of potentially exposed populations completes the characterization of the exposure setting at the site. This determination is significant because potential receptor populations could vary at different sites and because an exposure scenario is not complete if it is not reasonable to conclude that receptor populations in the vicinity of the site are subject to potential exposures. Evaluation of potentially exposed human populations is performed for distinct land-use conditions including current land use and future land use. The evaluation of potentially exposed populations of ecological receptors includes no land-use distinction.

5.1.4.1 Critical Subpopulations

According to the Risk Assessment Guidance for Superfund Volume I Human Health Evaluation Manual (Part A) (EPA 1989a), a baseline risk assessment must identify subpopulations of potential concern that could be at increased risk from radionuclide or chemical exposure from

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TABLE 5-1
POTENTIAL SOURCES OF CONTAMINANTS AT THE FEMP[a]

Operable Unit 1	Operable Unit 2	Operable Unit 3	Operable Unit 4	Operable Unit 5
- Waste Pits 1-6 - Clearwell - Burn Pit - Berms - Liners	- Fly Ash Piles - Southfield Disposal Areas - Lime Sludge Ponds - Solid Waste Landfill - Berms - Liners	- Production Area - Production-Associated Facilities/ Equipment - Structures - Equipment - Utilities - Drums - Tanks - Effluent Lines - K-65 Transfer Line - Wastewater Treatment Facilities - Fire Training Facilities - Scrap Metal Piles - Coal Pile - Feedstocks - By-Products - Products - Thorium Inventory - Biodenitrification Surge Lagoon	- K-65 Silos (Silos No. 1 and No. 2) - Metal Oxide Silo (No. 3) - Silo No. 4 - Decant Tank System - Berms	- All Contaminated Surface and Subsurface Soil Not Otherwise Associated with Other Operable Units - Perched Groundwater - Aquifer - Surface Water - Sediments - Flora and Fauna

[a] Each Operable Unit includes soils within the operable unit boundary (except Operable Unit 3) and water encountered during remediation.

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increased sensitivity, behavior patterns, and/or current or past exposures from other sources. These populations include infants and children, the elderly, pregnant and nursing women, individuals with chronic illnesses, and individuals previously exposed to chemicals or radionuclides during occupational activities or by residing in industrial areas. The current subpopulations of potential concern within five miles of the FEMP are identified below and are listed by the categories suggested by the EPA (1989a). The information presented on sensitive subpopulations covers the area within five miles of the FEMP and covers the area within between three and four miles of the leading edge of the South Plume. Within this distance from the South Plume the population difference based on 1990 census data is negligible and the descriptions of potential sensitive subpopulations are essentially the same. Subpopulations of potential concern will be identified in RI/FS risk assessments using 1990 census data.

- Schools: No schools are located within one mile of the FEMP. Three school districts provide public education from kindergarten through high school for children living within five miles of the FEMP. These are Northwest, Ross, and Southwest school districts. The 1989-90 total enrollment in the six schools from these districts within five miles of the FEMP was 3,316.

- Daycare Centers: No daycare facilities are located within one mile of the FEMP. Two daycare centers operate within the study area: (1) Ross County Day Nursery, with an average enrollment of 126 students per day and a total weekly enrollment of 180, is located north of the intersection of SR 128 and US 27 about two and one-half miles northeast of the center of the FEMP, (2) Venice Presbyterian Pre-School, with an average daily enrollment of 30 and a total weekly enrollment of 110, is located in the village of Venice (Ross) approximately two miles northeast of the center of the FEMP.

- Hospitals, Nursing Homes, and Retirement Communities: No care facilities of these types operate within five miles of the FEMP.

- Residential Areas with Children: In 1988, approximately 58 adults and 29 children were residing within one mile of the FEMP. Most of the residences within five miles of the FEMP are scattered and reflect the agricultural setting of the area. Population concentrations include Ross, Harrison, Shandon, Fernald, New Haven, New Baltimore, and one large trailer park. An estimated 8,140 children lived within five miles of the center of the FEMP in 1988.

- Commercial and Recreational Fisheries: No commercial fisheries operate within five miles of the center of the FEMP. Recreational fishing occurs on Whitewater Lake of the Miami Whitewater Forest Park. This heavily stocked lake lies completely within five miles of the FEMP. The Great Miami River supports no commercial fisheries in the vicinity of the FEMP, but recreational fishing occurs downstream of the FEMP. A fishing advisory for PCBs in bottom-feeding fish was issued in 1989 by the Ohio Department of Health based on data collected by Ohio EPA.

- Major Industries Using Chemicals: No industrial facilities are located within one mile of the center of the FEMP. Two companies located within two miles of the FEMP center, Ruetgers-Nease Chemical Company and Albright and Wilson, store and handle chemicals. Collectively known as the Paddys Run Road Site, these facilities are classified as CERCLA sites, are listed on the Comprehensive Environmental Response Compensation and Liability Information System (CERCLIS), and are undergoing a state-led RI/FS. Proctor & Gamble has a research facility approximately two miles east of the FEMP which is listed on CERCLIS and has undergone a Screening Site Inspection by U.S. EPA. Employees at these facilities are only considered a sensitive subpopulation if they reside within five miles of the FEMP.

5.1.4.2 Potentially Exposed Populations Under Current Land Use

Several possible exposure scenarios will be evaluated in the baseline risk assessments to investigate current human health risks from the FEMP. These can be divided into two groups: those accounting for the effects of current access controls, and those that discount the effects of access controls.

Potential Exposures Assuming Current Access Controls Continue

The selection and subsequent assessment of the potentially exposed population groups assumes that current land use of FEMP property will continue until remediation activities end, at which time active security controls will be discontinued. Scenarios incorporating the effects of custodial control of the property on off-property individuals include, but are not limited to:

- Visitor - This scenario investigates the exposures incurred by the activities of a regular visitor to the FEMP or one of its operable units who is not covered by the FEMP health and safety and radiation protection programs. An example of this scenario would be a delivery person making regular deliveries to the administration building in Operable Unit 3.

- Trespasser - This hypothetical scenario investigates the exposures incurred by the activities of a trespasser to the FEMP or one of its operable units who is not covered by the FEMP health and safety and radiation protection programs. Due to regular security patrols, this trespasser is assumed to be confined to areas near the property fenceline. Trespasser exposures will be evaluated, when appropriate, for individual operable units in the operable unit risk assessments and for the FEMP as a whole in the site-wide assessments.

- Exploring child - This hypothetical scenario supposes a child, aged 6 through 17, regularly ingests sediment while playing in Paddys Run. Exposures from sediments currently deposited along Paddys Run will be investigated as part of the Operable Unit 5 and site-wide risk assessments. Exposures from new sediment deposits resulting from future erosion of a soil/waste source will be evaluated during the assessment of the source's operable unit.

- Off-property farmer - This scenario presumes a farm family lives immediately adjacent to the FEMP property boundary. The exposure pathways included in this receptor scenario are expected to vary according to the location of the farm family in relation to the various soil/waste source areas. Typical activities evaluated might include growing food, tending livestock, and general farm work. These activities might result in radiation exposures from nearby soils; inhalation of gases, vapors and dust; and ingestion of water, dirt, and locally grown food such as crops, meat, and milk. In addition, Operable Unit 4 assessments might evaluate radiation exposures from the K-65 silos at the property boundary nearest the silos and include them in the farm family risk assessment. Conversely, gamma radiation from the K-65 silos would not be considered when evaluating off-property farm families located over the South Plume.

- On-property grazing - This scenario considers the risks associated with off-property use of animal products produced by cattle currently grazing on FEMP property. Receptors evaluated under this pathway may include off-property farmer families and other dairy/meat users.

Exposures from these scenarios will be presented separately during the FEMP risk assessments. They can also be combined in a summary presentation, if it is appropriate to do so.

Potential Exposures Assuming Current Access Controls Are Discontinued

The Amended Consent Agreement between DOE, OEPA, and EPA requires that "...each Baseline Risk Assessment shall include a scenario evaluating current conditions at the Site, assuming no further response actions and no institutional controls for the OU under consideration...". Therefore, each operable unit baseline risk assessment and the site-wide baseline risk assessment also will assess the risks for a hypothetical scenario that assumes environmental restoration of the property has ceased, and present access restrictions are discontinued. These evaluations consider only the current, unimproved condition of the property. Any activities requiring development time (i.e., home building, planting and harvesting crops, etc.) are addressed under future land use of the property (Section 5.1.4.3). Some potentially exposed population groups under these conditions might be:

- Visitor - This hypothetical scenario investigates the exposures incurred by the activities of a regular visitor to the FEMP or one of its operable units who is not covered by the FEMP health and safety and radiation protection programs. An example of this scenario would be a delivery person making regular deliveries to the property.

- Trespasser - Unrestricted trespassing on the FEMP property will be evaluated as part of the operable unit and site-wide baseline risk assessments. In this hypothetical scenario, individuals would regularly move about the property. They could be exposed to direct radiation, inhalation of resuspended soil, and ingestion of soil.

- Exploring child - This hypothetical scenario is identical to the previous (Trespasser) scenario except that the receptor is a child, aged 6 through 17.

- Off-property farmer - This hypothetical scenario presumes a farmer lives immediately adjacent to the FEMP property boundary. The exposure pathways included in this receptor scenario are expected to vary according to the location of the farm family in relation to the various soil/waste source areas. Typical activities evaluated might include growing food, tending livestock, and general farm work. These activities might produce radiation exposures from nearby soils; inhalation of gases, vapors and dust; and ingestion of water, dirt, and locally grown food such as crops, meat, and milk. Since access to the property is unrestricted for this scenario, additional pathways will be considered when evaluating the hypothetical risks to these nearby farm families. For example, radiation exposures from the K-65 silos to an individual tending cattle could be evaluated near the silos and included in the farm family risk assessment. Because no crops are currently grown within the FEMP fenceline, off-property farmers could not eat contaminated vegetables from the property.

- On-property grazing - This hypothetical scenario considers the risks associated with using animal products produced by cattle currently grazing on FEMP property. These animals will have access to areas containing significant levels of contamination if access to the property is unrestricted.

- On-property building user - If the operable unit presently contains metal, concrete, or wooden buildings, one hypothetical scenario evaluated would be the immediate occupancy of one of these buildings by a family of hypothetical homesteaders. This family could ingest waste or contaminated soil, inhale resuspended dust, and be directly exposed to radiation. Because no crops are currently grown within the FEMP fenceline, these homesteaders could not eat contaminated vegetables from the property. The resident could use animal products from livestock and wild animals currently grazing on FEMP property.

- Hunter - Unrestricted hunting on the FEMP property will be evaluated as part of the Operable Unit 5 and site-wide baseline risk assessments. In this hypothetical scenario, individuals would regularly move about the property. They would use animal products from wild animals currently found on FEMP property. They could be exposed to direct radiation, inhalation of resuspended soil, and ingestion of soil.

Exposures from these scenarios will be presented separately during the FEMP risk assessments. They can also be combined by risk assessors, if it is appropriate to do so.

5.1.4.3 Future Land-Use Scenarios

Long-term risks to the public may be associated with the presence of hazardous substances remaining at the property in the future. These long-term risks will be evaluated under the baseline (no-action) and remedial action assessments using reasonable assumptions of future land uses at the property. Two future land use scenarios which will be considered during FEMP risk assessments are presented below:

- Resident farm family - Examination of past and present local land-use practices suggests that it is reasonable to assume FEMP land would revert to residential and agricultural uses in the future, after remedial activities cease. Thus, receptors could reside directly on former FEMP property, and sensitive subpopulations, such as children or elderly residents, could be exposed directly to contaminated soils, groundwater, surface water, or airborne emissions from unremediated on-property soils and waste areas as a result of natural or anthropogenic activities.

  This farm family scenario assumes a family resides on-property, eats food grown on-property, drinks water drawn from the Great Miami Aquifer beneath the site, inhales gases or dusts generated at the property, and ingests soil as a result of activities at the farm. Typical activities evaluated might include growing food, tending livestock, and general farm work. These activities might produce radiation exposures from nearby soils; dermal absorption through contact with contaminanted soil and water; inhalation of gases, vapors and dust; and ingestion of water, dirt, and locally grown food such as crops, meat, and milk. Risks to these hypothetical on-property receptors will be evaluated for the next 1000 years as part of a resident farm family scenario.

- Construction intruder - Home builders comprise a second group of receptors which may be exposed to on-property contamination in the future. This scenario is identified in this series of assessments as the construction intruder scenario. It consists of an individual digging a basement and well, and building a house on the property. These activities might produce radiation exposures from nearby waste/soil, dermal absorption through direct contact with waste/soil, inhalation of gases, vapors, and dusts, and inadvertant ingestion of soil. Completion of construction ends the exposure scenario. This individual can be either an on-property resident farmer, or an individual living off-property. Exposures to this receptor will be presented separately from other future exposures. They can also be combined with exposures from other scenarios, if appropriate.

Future off-property populations could be exposed as a result of transport of hazardous materials from the FEMP to off-property locations. In addition to on-property farm families, the long term risks to some of the potentially exposed human populations listed under current land use in Section 5.1.4.2 may also be evaluated.

Institutional Controls During Implementation of Remedial Action Alternatives

For FS alternatives other than the no-action alternative, current land use assumes restricted access to the vicinity of the remediation during implementation of an alternative. Evaluation of the short-term effectiveness criterion during implementation of a remedial alternative will be based on this land-use assumption. Health risks to off-property members of the public and workers on-property that are not covered by the FEMP approved health and safety and radiation protection plans will be assessed during implementation of remedial alternatives. Additional information on FS risk assessments is provided in Section 10.0.

5.1.4.4 Occupational Receptors

The work force at the FEMP will be divided into two groups for risk assessment purposes. One group will include only those workers involved in remediation activities. All other workers will be included in the second group. Table 5-2 lists the other workers in this second group.

In general, these other workers are adults, ranging in age from 18 to 65 years old. Workers spending significant time on the property are covered by a comprehensive health and safety program under which employee exposures are managed and recorded, as required by 29CFR1910 (Occupational Safety and Health Administration [OSHA]) and 10CFR20 (NRC 1991). The only workers on the property not covered by this program are contractors and delivery personnel who are admitted to the property for a limited duration. They are treated as members of the general public.

Remediation Workers

Remediation at the FEMP will involve operations that can produce short-term occupational exposures. Typically, each operation involving potential exposures will be identified, and the activities and locations producing the highest exposure will be used as the occupational RME scenario. Some of the factors to be considered when determining the occupational RME for each major type of operation are:

- Worker's proximity to the waste
- Any factors reducing worker exposure rates (engineering and administrative controls, personal protective apparel, etc.)
- Duration of exposure
- Type of exposure (airborne dust, dermal contact, direct radiation, etc.)

Generally, the types of short-term occupational exposures expected to dominate the occupational RME scenario at the FEMP are inhalation of resuspended dust, inhalation of radon and radon daughters, and irradiation by gamma emitters. Other exposure pathways will be considered, including dermal contact and inhalation of vapors. The parameters used to assess these potential exposure pathways will be specific to the occupational activity performed.

Nonremediation Workers

The exposures of FEMP employees not involved with remediation will be assessed under the FEMP Health and Safety Program (Table 5-2). This program stipulates that workplaces within the FEMP must be monitored if their exposure rates exceed a predetermined level. This level has been established by DOE Order 5480.11 and OSHA 29CFR1910.96 as being acceptable.

TABLE 5-2
OCCUPATIONAL RECEPTORS

	Baseline Current Land Use	Baseline Future Land Use	FS Alternatives
Remediation Worker	N	N	O,Y[a]
Permanent Employee Not Involved With Remediation	O,N	O,N	O,N
Temporary Employee Not Involved With Remediation	O,N	O,N	O,N
Contractor Not Involved With Remediation	O,N	O,N	O,N
Delivery Services/ Visitors	Y	Y	Y

N - No remediation under the baseline scenario, not evaluated.
O - Covered by Health and Safety Plan, not evaluated.
Y - Evaluated.

[a] Required for evaluation of short-term risks.

The only workers at the FEMP not considered by this Health and Safety Program are contractors and delivery personnel who are admitted to the property for a limited duration. (Most contractors are expected to comply directly with this program, or operate under a program comparable to the FEMP Health and Safety Program.) It is assumed that some delivery workers are not covered by the FEMP program, so their exposures to airborne contaminants and direct gamma radiation will be evaluated as part of the FEMP risk assessments.

5.1.5 Ecological Receptors

A complete discussion of potential ecological receptors at the FEMP can be found in Facemire et al. (1990). The following discussion is largely drawn from that report, with additional sources cited appropriately.

Plants

Typical grasses found on the FEMP include red fescue, Kentucky bluegrass, and timothy. Herbs include teasel, red and white clovers, and goldenrod. The dominant tree species in the pine plantations is white pine, and common trees in the deciduous and the riparian woodlands include white ash, American elm, eastern cottonwood, and box elder. The Inactive Fly Ash Disposal Area is dominated by American elm, eastern cottonwood, and black locust. Aquatic vascular plants and algae occur along Paddys Run and in wetland areas.

Terrestrial Animals

Examples of mammal species observed on the FEMP include white-tailed deer, red fox, raccoon, white-footed mouse, and muskrat. The most common birds breeding on site include the mourning dove, American robin, blue jay, and northern bobwhite. Raptor species observed on site are the northern harrier, red-shouldered hawk, Cooper's hawk, red-tailed hawk, and American kestrel. The eastern screech owl and great horned owl are also common. Amphibians and reptiles occurring on the FEMP include the American toad, spring peeper, eastern box turtle, and snapping turtle. Snakes observed on site include the eastern garter snake, black rat snake, and northern water snake. Approximately 130 insect families from 15 orders are represented in FEMP habitats. Leaf hoppers are abundant in all habitats, while less abundant groups include short-horned grasshoppers, leaf beetles, springtails, fruit flies, dark-winged fungus gnats, ants, bees, and wasps.

Aquatic Organisms

Paddys Run, the Great Miami River, and adjacent aquatic habitats harbor fish, amphibians, and a variety of benthic macroinvertebrates. The most common fish in Paddys Run are the bluntnose minnow, creek chub, and stoneroller minnow. Common macroinvertebrates include non-biting midges, mayflies, stoneflies, caddisflies, oligochaetes, and blackflies. Fish collected from the Great Miami River near the FEMP include gizzard shad, freshwater drum and carp (Miller et al. 1987, 1988, 1989). The flora of the Great Miami River include aquatic vascular plants and a variety of unicellular and filamentous algae (Miller et al. 1988; USGS 1974 to 1982).

5.2 CHARACTERIZATION OF POTENTIAL EXPOSURE PATHWAYS AT THE FEMP

Environmental transport and exposure mechanisms at the FEMP are introduced in this section. A simplified conceptual transport and exposure model for the site is presented in Figure 5-5. This model is based on work performed to date for the RI/FS at the FEMP. The model depictsthe site and its surrounding environment and consists of different types of contaminant sources, environmental transport pathways, exposure mechanisms, and potential receptors.

5.2.1 Potential Water Exposure Pathways

The transport of contaminants from a source to groundwater begins with the infiltration of precipitation into a source area containing waste or contaminated soil, percolation of water through this matrix, and dissolution of contaminants by the water. This percolating water could carry contaminants downward through the source volume. In the event that the source volume allows the water to escape, the seepage could carry contaminants through the unsaturated zone below. Ultimately the seepage could reach the aquifer. Alternatively, the source may be deep enough to be in direct contact with perched groundwater. Groundwater can return to the surface environment in one or more of the following routes: through a seep or surface outcrop, by direct discharge to the Great Miami River or Paddys Run, or by being drawn to the surface as well water.

Transport of contaminants to surface water bodies, such as streams and rivers, is initiated by the runoff of precipitation over waste units and contaminated soils. This runoff erodes the soil/waste and carries the suspended and dissolved contaminants away from the source. The contamination in open waste pits also could contribute to surface water contamination if the open pits overflow during a storm. As the surface runoff event subsides, sediments are deposited in low flow drainage features, such as Paddys Run, the Storm Sewer Outfall Ditch, standing water areas, and wetlands. Large runoff events, or a series of small ones, can move this sediment downstream to the Great Miami River.

Water exposure pathways could exist for groundwater or for surface water. The aquifer is a potential source of water for residential, agricultural, and commercial use. Two commercial facilities proximal to the FEMP use groundwater for industrial purposes and nearby residents use it for agricultural purposes. Water in the Great Miami River is also a potential source of water for residential use, agricultural use, and commercial use. The river is the only potential surface 90 water supply in the area that could feasibly provide water in appropriate quantities on a consistent basis. Water exposure pathways are considered separately for groundwater and surface water as the primary source.

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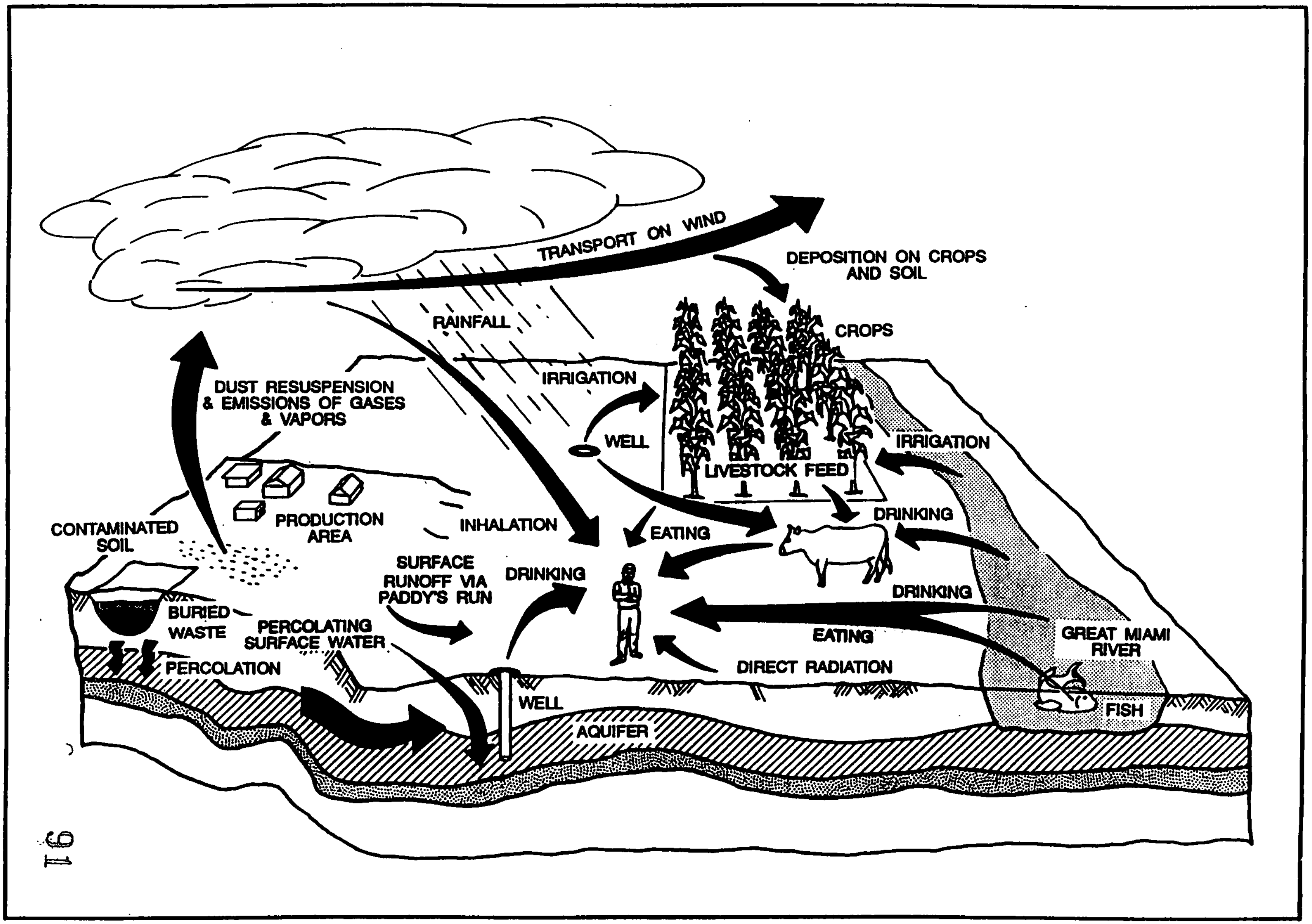

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FIGURE 5-5 SIMPLIFIED CONCEPTUAL SITE TRANSPORT/EXPOSURE MODEL FOR THE FEMP

Receptor exposures include exposures to contaminated water used as drinking water, water for irrigating food crops, water for irrigating feed crops for livestock, and drinking water for livestock. In addition, consumption of fish found in contaminated water can result in exposure. These water exposures involve contamination of the food chain. Additional exposures to contaminated water that do not involve the food chain include direct contact with contaminated water (potential dermal absorption of contaminants), incidental ingestion of surface water while swimming, and inhalation and dermal exposure to gases and volatile organic compounds released from contaminated water during household use or agricultural use such as showering or spray irrigating.

Ecological receptors may also be exposed to constituents in groundwater and surface waters. Exposure of aquatic organisms to constituents in groundwater could occur indirectly by seepage of groundwater into surface waters or by extraction of groundwater by humans, with subsequent release to surface waters. Potential pathways by which ecological receptors could be exposed to contaminants in surface water include ingestion, direct exposure of aquatic organisms, and indirect exposure via food chain uptake.

5.2.2 Potential Air Exposure Pathways

The transport of contaminants from a source to the air begins with either the resuspension of contaminated particulates on exposed surfaces or the emission of contaminants from a source area. Airborne contaminants are subsequently dispersed in the environment by winds and deposited on exposed surfaces, such as surface soil, plants, and structures. Contaminated surface soils, inactive production facilities, and open waste units such as the waste pits provide sources of contaminants on exposed surfaces that could be resuspended and transported elsewhere in the environment. Gaseous or volatile contaminants (such as radon or acetone) could be released to the air from a contained source area such as waste materials inside the silos, the solid waste landfill, or inside covered waste storage pits. Airborne isotopes of radon (Rn-222, Rn-220, Rn-219) may pose a potential risk in buildings at the site, especially in buildings that are contaminated with parent radionuclides of radon or in buildings used to store drums of material that contain the parent radionuclides. Risks from radon and its daughters will be assessed if parent radionuclides of radon are present or suspected.

Unique source-to-air relationships exist at the FEMP. For example, the K-65 silos release significant quantities of radon gas to the air. The radon gas is produced inside the silos by the decay of radium contained in the waste material. Baseline risk assessments also include scenarios where currently contained sources lose containment with the passage of time.

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Exposures occur as receptors are exposed to airborne contaminants or after airborne contaminants are deposited on exposed surfaces. The primary exposure to airborne contaminants results from inhalation of these contaminants. After airborne contaminants deposit on exposed surfaces, receptors may also be exposed to penetrating radiation from radiological contaminants. Less direct routes of exposure center on food pathways. Particle deposition on plants and soil and root uptake by food crops and animal feed allow contaminants to enter agricultural products. Exposures result when humans ingest these contaminated products.

5.2.3 Potential Soil Exposure Pathways

Exposures could occur after contaminants associated with the FEMP are transported to the soil via air transport and deposition, spills, irrigation, or waste storage/disposal. Human receptors could be exposed via incidental ingestion of contaminated soil, direct external contact with contaminated soil, direct radiation from the soil, consumption of produce grown on contaminated soil, and consumption of meat and milk from livestock that ingest contaminated soil or plants. Thus, contaminants transported to the soil could enter the food chain through the surface soil.

In addition, exposures could occur via contact with other media contaminated through erosive forces or water percolation and leaching of contaminants from the soil to these other media. Thus, the contaminated soil also serves as a potential source area with transport to other exposure media.

Potential pathways by which ecological receptors could be exposed to FEMP constituents in soils include: uptake of constituents from soils by plants; direct exposure of plants and animals to contaminated soils, including direct radiation; incidental ingestion by grazing animals; future exposure to constituents eroded by runoff; and indirect exposure via food chain uptake.

5.2.4 Potential Sediment Exposure Pathways

Exposures could occur after contaminants are transported to sediments from other source media such as by erosion by runoff and transport to surface waters such as Paddys Run, the Storm Sewer Outfall Ditch, and the Great Miami River. Contaminants introduced into these surface waters could subsequently settle and become incorporated into the stream bed. Human exposure could occur from incidental ingestion of contaminated sediment, from direct radiation, and from dermal contact with contaminated sediment.

Potential pathways by which ecological receptors could be exposed to FEMP constituents in sediments include: uptake of constituents by aquatic plants; direct exposure of aquatic plants and animals, including direct radiation exposure; and indirect exposure via food chain uptake. Ecological receptors could also be exposed to FEMP constituents in waste units via direct exposure of terrestrial animals to wastes, direct radiation, and for solid wastes, pathways similar to soils.

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5.3 SELECTION OF EXPOSURE PATHWAYS

Once all potential exposure pathways have been identified, it is desirable to select the potentially significant ones for a more detailed evaluation. EPA guidance for performing risk assessments (EPA 1989a) suggests eliminating an exposure pathway from detailed analysis when there is sound justification for elimination (e.g., based on the results of a screening analysis). EPA risk assessment guidance offers examples of justification for eliminating exposure pathways, including:

- "The exposure resulting from the pathway is much less than that from another pathway involving the same medium at the same exposure point."
- "The potential magnitude of exposure from a pathway is low."
- "The probability of the exposure occurring is very low and the risks associated with the occurrence are not high." (EPA 1989a)

An exposure pathway will be selected for detailed evaluation only if it is a complete exposure pathway or, in the case of a future pathway, potentially complete. A complete exposure pathway generally comprises four basic components:

- A source of contaminants
- A mechanism(s) for transporting contaminants to the point of receptor exposure
- A receptor present at a point where contaminants are present
- A mechanism for exposure of the receptor to the contaminants

An exposure pathway will be eliminated from quantitative evaluation if any of the four components is determined to be absent (Figure 5-6). A degree of reasonableness will be used when deciding whether the last two components are present (a receptor at a point where there are contaminants and a mechanism by which the receptor is exposed).

There are exceptions to this process for direct exposure pathways, such as exposure to penetrating radiation emitted from a radionuclide source. In such a case there is no need to consider a transport mechanism for exposure to occur. This screening process will be applied to every potential exposure pathway identified. This process will eliminate unreasonable pathways and focus on the list of potential exposure pathways selected for evaluation in the risk assessment.

The FEMP contains a large number of potential exposure pathways. Each exposure pathway consists of a source of contamination, a transport pathway or exposure mechanism, and a receptor. Table 5-3 lists these potential pathways, categorized by source and environmental medium. These pathways were screened for each operable unit land-use scenario using EPA

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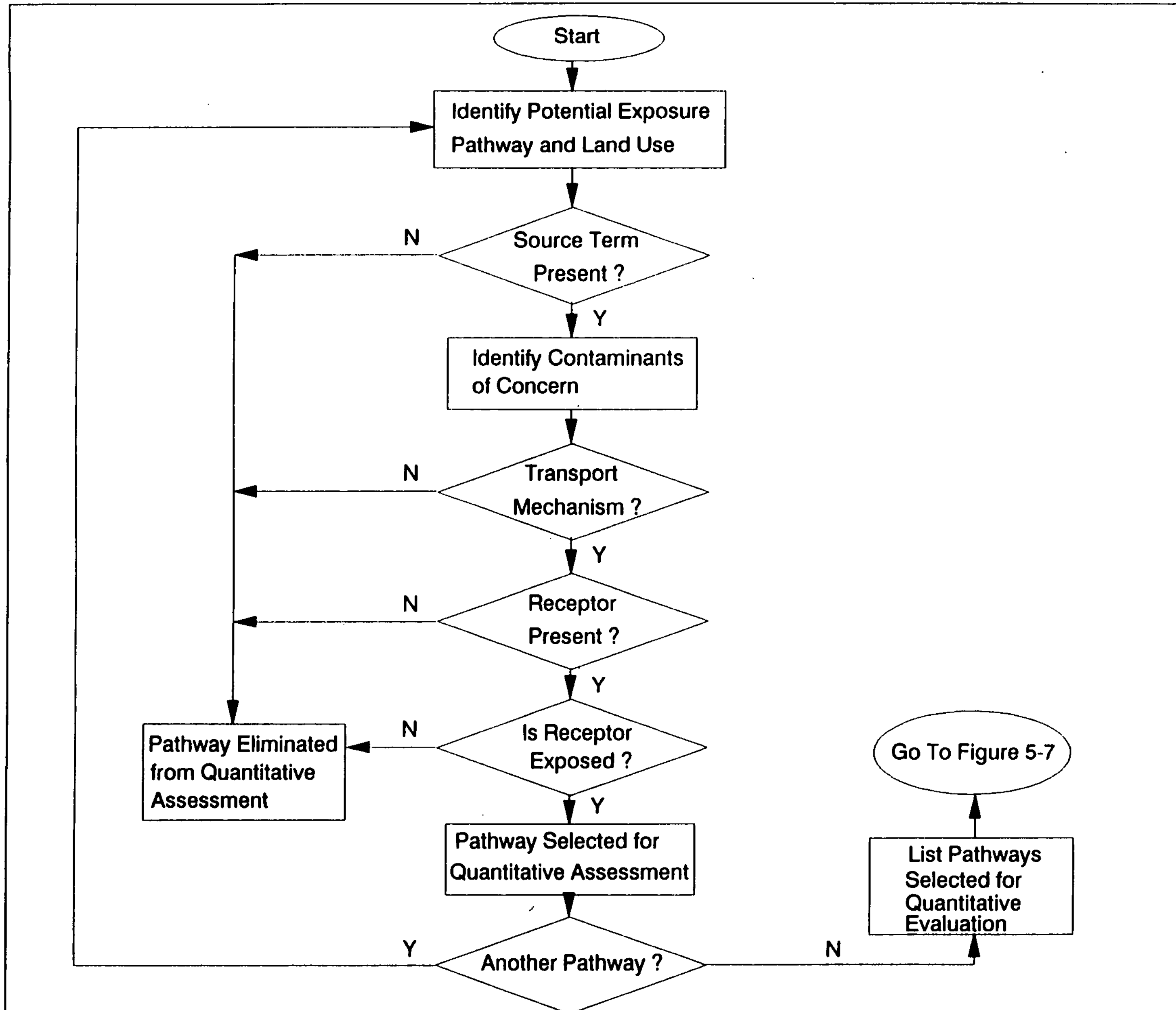

Figure 5-6 IDENTIFICATION AND SELECTION OF EXPOSURE PATHWAYS

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Table 5-3
Summary of Potential Pathways Evaluated in Assessment of Long-term Risks at the FEMP[a]

Id No	Exposure Pathways			Operable Unit 1 Scenario			Operable Unit 2 Scenario			Operable Unit 3 Scenario			Operable Unit 4 Scenario			Operable Unit 5 Scenario			Site-Wide Operable Unit Scenario		
	Source	Pathway	Exposure Media or Mechanism	1[b]	2[c]	3[d]	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
1	Soil/Waste	Foliar deposition	Crops	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•
2	Soil/Waste	Groundwater-irrigation	Crops			•			•			•			•			•			•
3	Soil/Waste	Root uptake	Crops			•			•			•			•			•			•
4	Soil/Waste	Surface water-irrigation	Crops			•			•			•			•			•			•
5	Soil/Waste	Surface soil	Dermal contact		•	•		•	•	•	•	•		•	•	•	•	•	•	•	•
6	Soil/Waste	Surface water-recreation	Dermal contact			•			•			•			•			•			•
7	Soil/Waste	Surface water-sediment	Dermal contact			•			•			•			•			•			•
8	Soil/Waste	Surface water-sediment	Direct ingestion			•			•			•			•			•			•
9	Soil/Waste	Surface soil	Direct ingestion		•	•		•	•	•	•	•		•	•	•	•	•	•	•	•
10	Soil/Waste	Groundwater-well	Domestic water			•			•			•			•			•			•
11	Soil/Waste	Surface water	Domestic water			•			•			•			•		•	•		•	•
12	Soil/Waste	Groundwater-well	Drinking water			•			•			•			•			•			•
13	Soil/Waste	Surface water	Drinking water			•			•			•			•		•	•		•	•
14	Soil/Waste	Surface water	Fish			•			•			•			•			•			•
15	Soil/Waste	Surface water-recreational	Incidental ingestion		•	•		•	•			•			•		•	•		•	•
16	Soil/Waste	Emission of gases to air	Inhalation	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•
17	Soil/Waste	Particulate resuspension	Inhalation	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•
18	Soil/Waste	Cloud immersion	Irradiation	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•
19	Soil/Waste	Proximal exposure	Irradiation		•	•		•	•	•	•	•	•	•	•	•	•	•	•	•	•
20	Soil/Waste	Surface water-recreation	Irradiation		•	•		•	•			•			•		•	•		•	•
21	Soil/Waste	Surface water-sediment	Irradiation		•	•		•	•			•			•		•	•		•	•
22	Soil/Waste	Ingestion by livestock	Meat & Milk		•	•		•	•		•	•		•	•	•	•	•	•	•	•
23	Soil/Waste	Contaminated Foliage	Meat & Milk	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•	•
24	Soil/Waste	Stock water	Meat & Milk			•			•			•			•		•	•		•	•

a Pathways may be evaluated either qualitatively or quantitatively. See Section 5.3.1 for numbered pathway descriptions.
b Scenario 1 - Current land use practices.
c Scenario 2 - Current land use, without access controls.
d Scenario 3 - Projected future land use practices.

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Table 5-3
(Continued)

Id No	Exposure Pathways: Source	Pathway	Exposure Media or Mechanism	Operable Unit 1 Scenario: 1[b]	2[c]	3[d]	Operable Unit 2 Scenario: 1	2	3	Operable Unit 3 Scenario: 1	2	3	Operable Unit 4 Scenario: 1	2	3	Operable Unit 5 Scenario: 1	2	3	Site-Wide Operable Unit Scenario: 1	2	3
25	Structures	Salvage/dust	Ingestion								•	•								•	•
26	Structures	Salvage/dust - gases	Inhalation								•	•								•	•
27	Structures	Salvage/Irradiation	Irradiation								•	•								•	•
28	Groundwater	Irrigation of crops	Crops													•	•	•	•	•	•
29	Groundwater	Well water	Domestic water													•	•	•	•	•	•
30	Groundwater	Well water	Drinking water													•	•	•	•	•	•
31	Groundwater	Irrigation of forage	Meat & Milk													•	•	•	•	•	•
32	Groundwater	Stock water	Meat & Milk													•	•	•	•	•	•
33	Surface water	Irrigation of crops	Crops			•			•							•	•	•	•	•	•
34	Surface water	Recreation	Dermal contact		•	•		•	•							•	•	•	•	•	•
35	Surface water	Water use	Domestic water			•			•							•	•	•	•	•	•
36	Surface water	Water use	Drinking water			•			•							•	•	•	•	•	•
37	Surface water	Fishing	Fish			•			•							•	•	•	•	•	•
38	Surface water	Recreation	Incidental ingestion		•	•		•	•							•	•	•	•	•	•
39	Surface water	Recreation	Irradiation		•	•		•	•							•	•	•	•	•	•
40	Surface water	Irrigation of forage	Meat & Milk			•			•							•	•	•	•	•	•
41	Surface water	Stock water	Meat & Milk		•	•		•	•							•	•	•	•	•	•
42	Sediment	Recreation	Dermal contact		•	•		•	•					•	•	•	•	•	•	•	•
43	Sediment	Recreation	Direct ingestion		•	•		•	•					•	•	•	•	•	•	•	•
44	Sediment	Recreation	Irradiation		•	•		•	•					•	•	•	•	•	•	•	•

a Pathways may be evaluated either qualitatively or quantitatively.
b Scenario 1 - Current land use practices.
c Scenario 2 - Current land use, without access controls.
d Scenario 3 - Projected future land use practices.

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guidance presented earlier. Pathways selected for detailed analysis during the FEMP RI/FS process are marked with a bullet ("•") in the appropriate row and column of Table 5-3. This matrix will be reviewed for accuracy and completeness during each RI/FS risk assessment.

Exposure pathways are grouped in Table 5-3 according to five source types. The sources are divided among operable units according to the definitions of operable units presented in Section 1.7 and the modified Consent Agreement. For example, groundwater currently located under the Waste Disposal Area will be treated as a source in the Operable Unit 5 and site-wide assessments. Exposures attributable to that source will be assessed only in those assessments. Operable Unit 1 will assess neither current nor future exposures from this groundwater source, but will assess exposures from any additional groundwater originating from the soil/waste sources in Operable Unit 1.

5.3.1 Soil/Waste Exposure Pathways

These pathways start with soil or waste materials as the ultimate source of the postulated exposures. This group contains the largest number of potential exposure pathways because of the large number of source types and transport mechanisms present at the site. Each pathway is listed in Table 5-3 and described below:

1. Ingestion of crops contaminated by foliar deposition of soil/waste. This pathway assumes aerial suspension of exposed soil/waste, followed by foliar deposition onto plants. These plants are later harvested and eaten by humans. This pathway will be evaluated for both current and future scenarios at the FEMP.

2. Ingestion of crops contaminated by irrigation with groundwater contaminated by soil/waste. This pathway postulates future contamination of groundwater by interactions with the soil/waste. This water could migrate to the receptor's location, where it may be pumped to the surface and used to irrigate food crops. This irrigation results in foliar deposition onto plants and uptake of contaminants by plant roots. These plants are later harvested and eaten by humans. This pathway will be evaluated for all future scenarios at the FEMP. See pathway 28 for crop ingestion exposures from presently contaminated groundwater.

3. Ingestion of crops contaminated by root uptake from soil/waste. This pathway postulates the direct contact of crop plant roots with contaminated soil/waste. The roots take up contaminants, and these plants are later harvested and eaten by humans. Since no crops currently exist on FEMP property, this pathway will be evaluated only for future scenarios.

4. Ingestion of crops contaminated by irrigation with surface water contaminated by soil/waste. This pathway assumes future contamination of surface water by the soil/waste. This water is used to irrigate food crops. Irrigation results in foliar deposition onto plants and uptake of contaminants by plant roots. These plants are later harvested and eaten by humans. Since no crops currently exist on FEMP property, this pathway will be evaluated only for future scenarios. See pathway 33 for crop ingestion exposures from presently contaminated surface water.

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5. Dermal contact with soil/waste. This pathway presumes a receptor can come into direct contact with the soil/waste, either on-property or off-property, now or in the future. Once in direct contact, uptake of certain contaminants may occur by dermal absorption. This pathway will be assessed for all scenarios which allow unrestricted or special access to potentially contaminated areas. Receptors which have special access to portions of the property may include (but are not necessarily limited to) delivery personal (OU3), and a farmer tending cows grazing on-property (OU5).

6. Dermal contact while swimming in Great Miami River water contaminated by soil/waste. This pathway postulates future contamination of surface water by soil/waste. This water then drains into the Great Miami River. A receptor then swims in this water. Once in direct contact with the water, uptake of certain contaminants may occur by dermal absorption through the receptor's skin and mucus membranes. This pathway will be assessed for future scenarios. See pathway 34 for exposures from dermal contact while swimming in presently contaminated surface water.

7. Dermal contact with sediment eroded and transported from soil/waste by surface water runoff. This pathway presumes surface deposits of soil/waste will be eroded in the future and transported as sediment to Paddys Run, the Storm Sewer Outfall Ditch, and the Great Miami River. Receptors using these waterways for recreation may then inadvertently ingest this sediment. See pathway 42 for dermal exposures from sediments presently in surface water.

8. Direct ingestion of sediment eroded and transported from soil/waste by surface water runoff. This pathway presumes surface deposits of soil/waste will be eroded and transported as sediment to Paddys Run, the Storm Sewer Outfall Ditch, and the Great Miami River. Receptors using these waterways for recreation may then inadvertently ingest this sediment. See pathway 43 for a description of the pathway to be used when estimating exposures from ingesting sediments presently in surface water.

9. Direct ingestion of soil/waste. This pathway assumes a receptor can come into direct contact with the soil/waste, either on-property or off-property, now or in the future. During the receptor's period of contact, the individual inadvertently ingests a small amount of soil/waste. This pathway will be assessed for all scenarios which allow unrestricted or special access to potentially contaminated areas. Receptors which have special access to portions of the property may include (but are not necessarily limited to) delivery personal (OU3), and a farmer tending cows grazing on-property (OU5).

10. Domestic use of groundwater contaminated by soil/waste. This pathway postulates contamination of groundwater in the future by interactions with soil/waste. This water migrates to the receptor location, where it is pumped to the surface and used for domestic (non-drinking) water. This allows exposures from dermal contact with the contaminated water (showering) and inhalation of constituents released from the water by off-gassing of volatile organic vapors and gases such as radon. See pathway 29 for a description of the pathway to be used when estimating exposures from ingesting presently contaminated groundwater.

11. Domestic use of surface water contaminated by soil/waste. This pathway postulates future contamination of surface water by soil/waste. This water then drains into the Great Miami River where it is treated and used for domestic (non-drinking) water. This allows exposures from dermal contact with the contaminated water (showering) and inhalation of constituents released from the water by off-gassing of volatile organic vapors and gases such as radon. See pathway 35 for a description of the pathway to be used when estimating exposures from using water currently available in the Great Miami River.

12. Ingestion of groundwater contaminated by soil/waste. This pathway postulates contamination of groundwater in the future by interactions with soil/waste. This water migrates to the receptor location, where it is pumped to the surface and used as a supply of drinking water. See pathway 30 for a description of the pathway to be used when estimating exposures from ingesting presently contaminated groundwater.

13. Ingestion of surface water contaminated by soil/waste. This pathway postulates future contamination of surface water by soil/waste. This water then drains into the Great Miami River where it is treated and used for municipal drinking water. This pathway will be assessed for future scenarios. See pathway 36 for exposures from drinking water currently available in the Great Miami River.

14. Ingestion of fish raised in surface water contaminated by runoff from soil/waste. This pathway assumes surface water is contaminated by soil/waste deposits in the future. This water drains into bodies of surface water containing food fish. These fish are caught and eaten. See pathway 37 for exposures from eating fish taken from the present Great Miami River under current conditions.

15. Incidental ingestion of surface water contaminated by soil/waste. This pathway postulates future contamination of surface water by soil/waste, which drains into the Great Miami River. A receptor then accidentally ingests a small amount of this water while swimming. This pathway will be assessed for future scenarios. See pathway 38 for exposures from dermal contact while swimming in presently contaminated surface water.

16. Inhalation of gases emitted from soil/waste. This pathway postulates the emission of gases such as radon and volatile organic vapors from soil/waste. This is followed by their transportation through the soil and air to the vicinity of the receptor (either indoors or outdoors). The receptor then inhales these gases. The pathway will be analyzed for both current and future scenarios.

17. Inhalation of suspended particulates from soil/waste. This pathway assumes aerial suspension of exposed soil/waste, and subsequent transport through the air as dust to the vicinity of the receptor. The outdoor receptor inhales this dust. The pathway will be analyzed for both current and future scenarios.

18. Radiation exposures during immersion in a cloud of gas produced by soil/waste. This pathway assumes soil/waste produces radioactive gases such as radon-222. 100 These gases are either emitted in the immediate vicinity of a receptor (e.g. in a home), or are transported by atmospheric processes to the vicinity of the receptor.

The receptor receives an exposure by direct radiation from the radionuclides in the gas cloud. This pathway will be considered for both current and future scenarios.

19. <u>Proximal exposures via direct radiation from soil/waste.</u> This pathway presumes a receptor can approach the location of the soil/waste, either on-property or off-property, now or in the future. The receptor receives an exposure by direct radiation from the radionuclides in the soil/waste. This pathway will be assessed for all scenarios which allow either unrestricted or special access to potentially contaminated areas, and those scenarios involving the K-65 silos. Receptors which have special access to portions of the property may include (but are not necessarily limited to) delivery personal (OU3) and a farmer tending cows grazing on-property (OU5).

20. <u>Radiation exposures during immersion in surface water contaminated by future interactions with soil/waste.</u> This pathway postulates future contamination of surface water by soil/waste. This water then drains into the Great Miami River, where swimmers may then be exposed by direct radiation from radionuclides dissolved or suspended in this water. This pathway will be evaluated for future scenarios. See pathway 39 for a presentation of the exposure pathway describing immersion exposures from currently contaminated surface water.

21. <u>Radiation exposures from sediment formed by future interactions with soil/waste.</u> This pathway assumes surface deposits of soil/waste will erode and subsequently be transported to local water bodies such as Paddys Run, the Storm Sewer Outfall Ditch, and the Great Miami River. Receptors using these waterways for recreation may then be exposed by direct radiation from radionuclides in this sediment. See pathway 44 for a presentation of the exposure pathway describing irradiation from currently contaminated deposits of sediment.

22. <u>Ingestion of meat and milk from livestock ingesting soil/waste.</u> This pathway presumes livestock can come into direct contact with the soil/waste. During grazing activities the animal inadvertently ingests soil/waste. The animal subsequently provides meat or milk that is used by a human receptor. This pathway will be assessed for all scenarios which allow animals to have unrestricted or special access to potentially contaminated areas. Animals which currently have special access to portions of the property include cattle grazing on-property (OU5).

23. <u>Ingestion of meat and milk from livestock eating forage contaminated by soil/waste.</u> This pathway assumes many transport mechanisms may be functioning at the same time to convey contaminants from soil/waste to the vicinity of the forage plant. The plant root may be physically located in the waste, foliar deposition of dust or irrigation water may take place, and/or root uptake of contaminated irrigation water may occur. Each of these transport pathways would be expected to increase the amount of contamination taken up by the plant over time. These plants are used as forage and stored feed by livestock. Meat and milk from these animals are later consumed by humans. Because of the air transport portion of this pathway, it will be evaluated for both current and future scenarios. See pathways 31 and 40 for presentations of pathways involving irrigation of forage using currently contaminated water.

24. Ingestion of meat and milk from livestock ingesting stock water contaminated by soil/waste. This pathway is actually a combination of two pathways. The first pathway postulates contamination of groundwater by interactions with the soil/waste. This water migrates to the receptor's location, where it is pumped to the surface and used to supply livestock with drinking water. The second pathway is identical to the first, except the second one assumes surface water (not groundwater) mobilizes and transports the contaminants from the waste to the receptor. The pathways are combined here because it seems likely that only one source of water (surface water or groundwater) will be used at one time. The transport pathway producing the highest exposures will be included in future scenarios. See pathway 32 and 41 for presentations of the exposure pathways describing the use of currently contaminated water sources for stock water.

5.3.2 Exposure Pathways Attributable to Salvage or Reuse of Structures

These pathways involve the use of existing contaminated structures as the ultimate source of the postulated exposures. The pathways are generally dependent on some degree of proximity to contaminants. They will often be combined with several of the soil/waste pathways listed in Section 5.3.2.1 to account for exposures produced by wastes contained within inactive process equipment or stored within a particular building. Three pathways listed in Table 5-3 are:

25. Ingestion of dirt during salvage or reuse of a structure. This pathway assumes buildings on the property are available for salvage or long-term reuse by an intruder. During salvage or other activities, the receptor may inadvertently ingest removable surface contamination. This pathway will be evaluated for all scenarios allowing unrestricted access to buildings on the site.

26. Inhalation of dust during salvage or reuse of a structure. This pathway postulates buildings on the property are available for salvage or long-term reuse by an intruder. During salvage or other activities, the receptor may inhale resuspended dust or other surface contamination. This pathway will be evaluated for all scenarios allowing unrestricted access to buildings on the site.

27. Irradiation during salvage or reuse of a structure. This pathway presumes buildings on the site are available for salvage or long-term reuse by an intruder. During salvage or other activities, the receptor may be irradiated by penetrating radiation from radionuclides found on the inner and outer surfaces of the facility. This pathway will be evaluated for all scenarios allowing unrestricted access to buildings on the site.

5.3.3 Exposure Pathways from Groundwater Sources

These pathways start with existing contaminated groundwater as the ultimate source of the postulated exposures. This group of pathways is considered during evaluation of exposures from currently contaminated media at the FEMP. Impacts associated with any additional production of contaminated groundwater will be assessed during the evaluation of the source of that contamination. For example, exposures from any existing contaminated groundwater under Operable Unit 1 are considered during evaluation of current scenarios in the Operable Unit 5 risk

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assessments. In addition, future migration of existing groundwater, and exposures associated with its subsequent use are considered during evaluation of future scenarios in the Operable Unit 5 assessments. Exposures attributable to any future contamination of groundwater by Operable Unit 1 wastes are considered during the Operable Unit 1 assessments. (See Section 5.3.2.1 for a description of pathways involving groundwater contaminated by future interactions with a soil/waste). The following exposure pathways involve currently contaminated groundwater, and are listed in Table 5-3:

28. Irrigation of crops using currently contaminated groundwater. This pathway assumes existing contaminated groundwater is used to irrigate food crops, either now or in the future. This irrigation results in foliar deposition of contaminated water onto plants and the uptake of contaminants by plant roots. These plants are later harvested and eaten by humans. This pathway will be assessed as for both current and future scenarios during the OU5 and site-wide assessments. See pathway 2 for a presentation of the pathway describing irrigation using groundwater contaminated by future interactions with soil/waste.

29. Use of existing groundwater as domestic water. This pathway postulates the use of existing contaminated groundwater. This water is pumped to the surface and used for domestic (non-drinking) water. This allows exposures from dermal contact with the contaminated water (showering) and inhalation of constituents released from the water by off-gassing of volatile organic vapors and gases such as radon. See pathway 10 for a presentation of the pathway describing exposures from groundwater contaminated by future interactions with soil/waste.

30. Use of existing groundwater as drinking water. This pathway postulates the use of existing contaminated groundwater. This water is pumped to the surface and used as a supply of drinking water. See pathway 12 for a description of the pathway describing exposures from groundwater contaminated by future interactions with soil/waste.

31. Ingestion of meat and milk from livestock fed forage irrigated with existing contaminated groundwater. This pathway assumes existing contaminated groundwater is used to irrigate feed crops. This irrigation results in foliar deposition of contaminated water onto plants and the uptake of contaminants by plant roots. These plants are used as forage and stored feed by livestock. Meat and milk from these animals are later consumed by humans. See pathway 23 for a presentation of the pathway describing exposures from irrigation water contaminated by future interactions with soil/waste.

32. Ingestion of meat and milk from livestock drinking existing contaminated groundwater. This pathway postulates the migration of existing contaminated groundwater and its subsequent use as drinking water for livestock. Meat and milk from these animals are later consumed by humans. See pathway 24 for a presentation of the pathway describing exposures from stock water contaminated by future interactions with soil/waste.

5.3.4 <u>Exposure Pathways from Existing Surface Water Sources</u>

These pathways start with existing sources of contaminated surface water as the ultimate source of the postulated exposures. Sources of potentially contaminated surface water near the FEMP are the Great Miami River, Paddys Run, and the Storm Sewer Outfall Ditch. Exposures from these surface water sources will be assessed in Operable Unit 5 and site-wide risk assessments. Some operable units contain ponds of standing water. These surface impoundments will be treated as reservoirs of contaminated surface water that can spread off property, or be accessed by an intruder in the future. Exposures from these surface water impoundments will be assessed during the evaluation of surface water pathways performed for their associated operable unit RI/FS. The following exposure pathways involving existing contaminated surface water are listed in Table 5-3:

33. <u>Ingestion of crops irrigated with currently contaminated surface water.</u> This pathway assumes existing contaminated surface water is used to irrigate food crops. This irrigation results in foliar deposition onto plants and uptake of contaminants by plant roots. These plants are later harvested and eaten by humans. See pathway 4 for a presentation of the crop ingestion pathway associated with surface water contaminated by future interactions with soil/waste.

34. <u>Dermal exposures from recreational use of Great Miami River water.</u> This pathway presumes a receptor swims in the Great Miami River. Once in direct contact with the water, uptake of certain contaminants may occur by dermal absorption through the receptor's skin and mucus membranes. See pathway 6 for a presentation of the dermal exposure pathway associated with surface water contaminated by future interactions with soil/waste.

35. <u>Domestic use of Great Miami River water.</u> This pathway postulates the use of treated Great Miami River water for domestic (non-drinking) purposes. This includes exposures from dermal contact with the contaminated water (showering) and inhalation of constituents released from the water by off-gassing of volatile organic vapors and gases such as radon. See pathway 11 for a description of the pathway to be used when estimating domestic exposures using surface water contaminated by future interactions with soil/waste.

36. <u>Using the Great Miami River as a source of drinking water.</u> This pathway postulates the use of treated Great Miami River water as a municipal drinking water source. See pathway 13 for a description of exposures associates with drinking surface water contaminated by future interactions with soil/waste.

37. <u>Ingestion of fish from the Great Miami River.</u> This pathway postulates the current existence of food quality fish in the Great Miami River. These fish are caught by humans and eaten. See pathway 14 for a presentation of the exposure pathway associated with fishing in surface water contaminated by future interactions with soil/waste.

38. <u>Incidental ingestion of Great Miami River water.</u> This pathway presumes a receptor accidentally ingests a small amount of untreated Great Miami River water while swimming. See pathway 15 for a description of the pathway involving accidental ingestion of surface water contaminated by future interactions with soil/waste.

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39. Immersion exposures by direct radiation from recreational use of existing contaminated surface water. This pathway presumes a receptor swims in the Great Miami River. Swimmers may then be exposed by direct radiation from radionuclides dissolved or suspended in this water. See pathway 20 for a presentation of the immersion exposure pathway associated with swimming in surface water contaminated by future interactions with soil/waste.

40. Ingestion of meat and milk from livestock fed forage irrigated with existing contaminated surface water. This pathway assumes existing reservoirs of contaminated surface water will be used to irrigate feed and forage. This irrigation results in foliar deposition of contaminated water onto plants and the uptake of contaminants by plant roots. These plants are used as forage and stored feed by livestock. Meat and milk from these animals are later consumed by humans. See pathway 23 for a presentation of the pathway describing exposures using irrigation water contaminated by future interactions with soil/waste.

41. Ingestion of meat and milk from livestock drinking existing contaminated surface water. This pathway presumes existing contaminated surface water will be used as drinking water for livestock. Meat and milk from these animals are later consumed by humans. See pathway 24 for a presentation of the pathway describing exposures from stock water contaminated by future interactions with soil/waste.

5.3.5 Exposure Pathways from Sediment Sources

These pathways begin with existing deposits of sediment as the ultimate source of the postulated exposures. This group of pathways will be evaluated as part of the Operable Unit 5 exposure evaluation of currently contaminated media at the FEMP. Impacts associated with any additional production of contaminated sediments will be assessed during the evaluation of the contamination's ultimate source. Each pathway is listed in Table 5-3 and described below:

42. Dermal contact with sediment. This pathway postulates the existence of contaminated sediments in Paddys Run, the Storm Sewer Outfall Ditch, and the Great Miami River. In addition, standing water in waste units can contain sediment. Receptors using these waterways for recreation may come into direct contact with this sediment. Once in direct contact, uptake of certain contaminants may occur by dermal absorption. See pathway 7 for a description of the exposure pathway associated with contacting sediment produced by future interactions with soil/waste.

43. Direct ingestion of sediment. This pathway postulates the existence of contaminated sediments in Paddys Run, the Storm Sewer Outfall Ditch, and the Great Miami River. In addition, standing water in waste units can contain sediment. Receptors using these waterways for recreation may then inadvertently ingest this sediment. See pathway 8 for a description of the exposure pathway associated with ingestion of sediment produced by future interactions with soil/waste.

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44. Proximal exposures via direct radiation from sediment. This pathway postulates the existence of contaminated sediment in Paddys Run, the Storm Sewer Outfall Ditch, and the Great Miami River. In addition, standing water in waste units can contain sediment. Receptors using these waterways for recreational uses may then be

exposed by direct radiation from radionuclides in this sediment. See pathway 21 for a description of the exposure pathway associated with irradiation from sediments produced by future interactions with soil/waste.

5.4 RME LOCATIONS

The RME location is the point or area where the reasonable maximum exposures to a real or potential receptor are calculated to occur. The RME location is determined from the overall RME scenario. Several factors influence the determination of this location, including contaminant concentration and toxicity, the degree of access receptors have to contaminated environmental media, land use on and around the site, and the lifestyles and physical attributes of the individuals likely to be exposed at that location. Each of these factors must be considered when determining the RME location. For example, it is generally true that the magnitude of an exposure is directly related to the concentration of a contaminant in environmental media. Thus a location possessing higher levels of contamination is more likely to produce higher exposures.

The extent to which a receptor has access to contaminated areas also influences the magnitude and type of exposure incurred. If a receptor has ready access to the location of the contaminated media, the resulting exposures will typically be higher than if the contamination was less accessible. For example, direct exposures to a receptor tilling soil will be greater if the contamination is on the surface than if the contamination is buried under several meters of soil. Current land-use restrictions with security measures (fences and routine patrols) are another example of how access to a contaminated area is presently limited or eliminated.

The lifestyle of the hypothetical receptor can influence the amount and types of exposures expected. Components of this lifestyle affecting the exposures incurred by the receptor include:

- The amount of local food ingested
- Time spent both indoors and outdoors by residents
- The amount of local water ingested
- The types of outdoor activities performed
- Behavior or physical attributes that would classify a receptor as a member of a critical population group, or increase the severity of the postulated exposure

For example, the lifestyle of a farmer residing on or near an operable unit would be expected to produce higher exposure rates than a transient intruder or a dweller working off-site.

5.4.1 Operable Unit RME Locations

The RME location for a given operable unit will be determined by first locating accessible areas on or near the operable unit that contain, or are likely to contain, elevated levels of contaminants of concern (Section 7.1). Next, information on local land use and population groups will be examined and a reasonable profile of the behavior and physical attributes of potential receptors will be developed. Potential intakes will then be quantified, for real or hypothetical individuals at each selected location, using information from the receptor's profile (Section 7.2).

The resulting exposures to the evaluated receptors will then be compared with each other, and the location producing the highest of these exposures will be designated as the RME location. In the case of multiple pathways and contaminants, the relative toxicities of the contaminants of concern will also be considered in the selection of the RME location. Table 5-4 lists the most probable RME locations, by operable unit, based on information available as of December 1, 1991. This table contains our current best estimate of RME locations and the dominant exposure pathways, and is subject to change upon completion of a baseline risk assessment. The pathways listed are examples of what the pathway producing the greatest amount of risk might be, and where the maximum exposure may be located. It should be noted that there is no intent to bias subsequent risk assessments towards the pathways and locations listed in the table. All reasonable pathways will be evaluated.

Potential influences from other operable units will not be considered when determining the operable unit RME. These impacts will be addressed by the Preliminary Site-Wide Baseline Risk Assessment, the Comprehensive Response Action Risk Evaluation accompanying each operable Unit FS, and by the Site-Wide Projected Residual Risk Assessment.

5.4.2 Site-Wide RME Locations

The reasonable maximum exposure location will be determined by first locating areas on or near the FEMP which contain elevated levels of contaminants of concern. The selection process is similar to the one used to determine the operable unit reasonable maximum exposure location (Section 5.4.1). These concentrations will be used to determine the location currently producing the reasonable maximum exposure.

Environmental fate and transport modeling will be used to predict concentrations when measured concentrations are not available, and for projections into the future. The many sources and transport mechanisms at the FEMP are expected to produce a complex matrix of interdependent effects requiring careful consideration. Thus, it will be necessary to account for the interactions of all operable units when predicting concentrations at the FEMP. 107

TABLE 5-4

EXAMPLES OF POSSIBLE REASONABLE MAXIMUM EXPOSURE (RME) LOCATIONS FOR THE BASELINE RISK ASSESSMENTS

	RME Individual	RME Location
Operable Unit 1		
Current situation		
with Controls	Off-site farmer	Fenceline, down gradient
w/o Controls	Off-site farmer[a]	Fenceline, down gradient[a]
Future scenario	Resident farmer	On site
Operable Unit 2		
Current situation		
with Controls	Child eating sediment	Paddys Run
w/o Controls	Off-site farmer[a]	Fenceline, down gradient[a]
Future scenario	Resident farmer	On site
Operable Unit 3		
Current situation		
with Controls	Adult eating soil	Fenceline, downgradient
w/o Controls	Off-site farmer[a]	Fenceline, downgradient[a]
Future scenario	Resident farmer	On site
Operable Unit 4		
Current situation		
with Controls	Off-site farmer	Fenceline at a point nearest to the silos
w/o Controls	Off-site farmer[a]	Fenceline at a point nearest to the silos[a]
Future scenario	Resident farmer	Immediately adjacent to silos
Operable Unit 5		
Current situation		
with Controls	Off-site farmer	Fenceline, downgradient
w/o Controls	Off-site farmer[a]	Fenceline, downgradient[a]
Future scenario	Resident farmer	Above South Plume area
Site-Wide Operable Unit		
Current situation		
with Controls	Off-site farmer	Fenceline, downgradient
w/o Controls	Off-site farmer[a]	Fenceline, downgradient[a]
Future scenario	Resident farmer	On site (Operable Unit 1)

[a] Includes hypothetical exposures incurred by receptors from unlimited trespassing on the site.

These interactions are expected to increase projected contaminant concentrations at locations where migrating contaminants from one or more operable units intersect static or migrating contaminants from another operable unit. The increased concentrations resulting from this intersection of contaminants may be sufficient to produce a site-wide RME at that location. This location could be synonymous with an existing operable unit RME location (Table 5-4), or it may be an entirely new location.

Operable unit interactions could also influence the selection of alternatives during the FS process. For example, a number of areas may be determined to be insufficiently protective of human health and the environment. An alternative designed to reduce the exposures from one location may also reduce exposures in a neighboring area. Thus a less intensive remedial alternative may be sufficient to reduce exposures to protective levels in the second area than would be indicated by studying the second area alone.

Potential risks from different operable units to hypothetical receptors at a specific location will be summed when assessing site-wide risks. The contribution of risks from any given operable unit or pathway to a selected receptor location may be minimal or nonexistent because the source locations and directions of contaminant migration from multiple operable units may be mutually exclusive at a receptor location.

5.5 QUANTITATIVE PATHWAY ASSESSMENT

Transport of contaminants along selected exposure pathways must be determined. This process is depicted in Figure 5-7. First, it must be determined whether available analytical results are sufficient to conduct the quantitative evaluation of the exposure pathway. If available data are sufficient, quantitative evaluation proceeds to the intake/exposure assessment step as depicted in Figure 5-7. If available data are deemed insufficient to perform the quantitative assessment, it becomes necessary to use a model to estimate a receptor exposure concentration or exposure level in lieu of analytical data.

In addition to the use of a model, it is also often appropriate to plan additional field investigations to obtain analytical data for quantitative evaluation of an exposure pathway. A decision to perform these additional field investigations is partially dependent on the potential magnitude of exposure that could be contributed by the exposure pathway and the degree of certainty estimated to be associated with the modeled results. A decision to model exposure concentration or exposure level leads to selection of the transport or source medium under consideration. Five choices are available in Figure 5-7; each is presented in detail in a referenced figure appearing in Section 6.0 of this addendum. The five distinct modeling pathways depicted in Figure 5-7 ultimately produce an estimated receptor exposure concentration or exposure level that is used in the intake/exposure assessment step depicted in the figure.

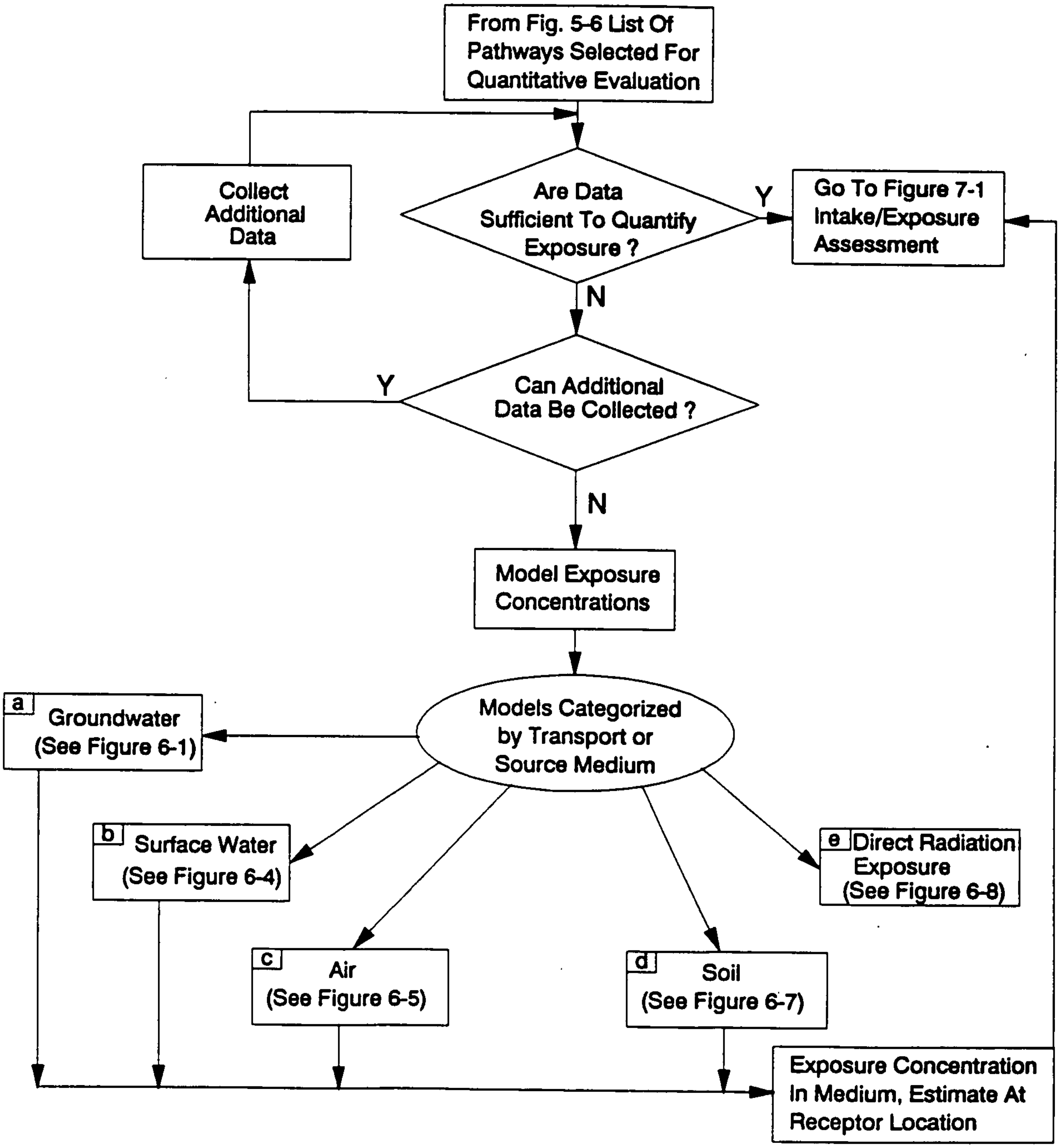

Figure 5-7 ASSESSMENT OF SELECTED EXPOSURE PATHWAYS

6.0

6.0 FATE AND TRANSPORT MODELING

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Fate and transport models are used to predict contaminant movement from source areas to receptor locations through various media. Used in conjunction with monitoring data, these models provide contaminant concentrations at potential exposure locations when measured contaminant concentration data are not available, such as for off-property locations or for future exposure predictions.

This section presents a description of the methodology used to quantitatively predict contaminant concentrations for use in FEMP risk assessments, including discussions of the fate and transport models to be used (Table 6-1) and their required data and default parameter values. In addition, the technical approach used to determine the appropriate model for each potential exposure assessment is discussed.

The models listed in Table 6-1 were obtained from a variety of references. This list is not all inclusive, and the final selection of models will be subject to EPA approval for each risk assessment. Each model was selected based on its appropriateness for a specific application in the risk assessment process, and the availability of input information required for the model. In general, these models provide estimates of contaminant concentrations in environmental media (e.g., air, water, or soil concentration) at a potential exposure point location. Cross-checking of the results of the different models will be performed where possible.

One goal of the modeling effort is to use input parameters and default values that are consistent with EPA recommendations. It is intended that input parameters and default values be used consistently for all models. Assumptions and parameters presented in this Work Plan Addendum may change, subject to EPA approval, as new information becomes available. Any changes from the default values presented here will be summarized in either text or tabular form in each risk assessment document.

Due to the large number of potential exposure pathways at the FEMP, the models are grouped by transport media. Models used to quantify fate and transport of contaminants in groundwater are presented in Section 6.1. Section 6.2 includes descriptions of surface water and sediment models. Section 6.3 presents the air transport models. Soil models are described in Section 6.4, while direct radiation exposure models are presented in Section 6.5. A discussion of sensitivity analyses and uncertainty analyses in risk assessments for the FEMP is given in Section 9.0.

TABLE 6-1
FATE AND TRANSPORT MODEL SUMMARY

Model Name	Model Type	Model Description/Use	Reference
AIRDOS-EPA Family of Codes	Air transport and dispersion model	Predicts air concentrations of contaminants off site (>100 m away).	EPA 1979, 1989d, Moore et al. 1989
Box Model	Air transport model	Predicts air concentrations of contaminants on site (<100 m away).	GRI 1988
EQ3NR and EQ6	Geochemical	Performs solubility speciation and reaction path calculations. Estimates leachate concentrations.	Wolery 1983, 1984
MICROSHIELD	Radiation shielding model	Calculates external gamma dose rate to an exposure point from a radiation source and intervening shield materials.	Grove 1988
MUSLE[a]	Soil loss by surface water erosion	Predicts annual soil loss to a stream based on event-specific rainfall.	EPA 1988c
ODAST	Vadose zone fate and transport	A one-dimensional model that evaluates fate and transport of remaining constituents in the vadose zone.	Javendel et al. 1984
PRESTO-EPA-CPG	Multiple pathway model	Calculates the CEDE[b] to a critical population group resulting from disposal of low-level radioactive waste.	EPA 1989d
RAECOM	Radon emanation	Predicts radon generation and radon flux emanating from waste and cover materials.	NRC 1984
RESRAD	Multiple pathway model	Calculates CEDE to a critical population group from residual radionuclide concentrations in soil.	DOE 1989
SESOIL	Vadose zone fate and transport model	Evaluates long-term environmental hydrologic, sediment and pollutant fate and transport.	EPA 1984a

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TABLE 6-1
(Continued)

Model Name	Model Type	Model Description/Use	Reference
ST1D	Vadose zone fate and transport	A one-dimensional model used for initial screening of constituent mobility in the vadose zone.	IT 1990a
SWIFT III	Solute fate and transport	SWIFT III is a 3-D finite-difference groundwater flow and solute transport code. Predicts flow and solute migration from the source through the groundwater system.	Geotrans 1987
USLE[c]	Soil loss by surface water erosion	Predicts annual soil loss to a stream based on area averaged annual rainfall.	EPA 1988c
Volatilization Model	Volatilization and dispersion model	Predicts volatilization and dispersion of VOCs[d].	GRI 1988

[a] MUSLE signifies Modified Universal Soil Loss Equation.
[b] CEDE signifies committed effective dose equivalent.
[c] USLE signifies Universal Soil Loss Equation.
[d] VOC signifies volatile organic compound.

6.1 GROUNDWATER TRANSPORT MODELING

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The two major sources of groundwater contamination at the FEMP involve leaching of solid contaminants from various sources and the percolation of contaminated liquids to the aquifer. The direct discharge of fluids to the vadose zone is possible from some of the sources within the Waste Storage Area, and it is possible that some ponds may seep directly into a perched zone of saturation, but leaching of waste solids and residual levels of contaminants in the soil is the most likely source of groundwater contamination for the rest of the site. Solid material itself does not contaminate groundwater directly because it will not migrate through the porous medium. Therefore, it is necessary for a liquid such as percolating soil water or groundwater to leach a portion of the available constituents from the solid material and transport the resulting leachate to the aquifer.

Migration of potential contaminants from FEMP sources through groundwater to a hypothetical receptor will be modeled as necessary for each risk assessment. Figure 6-1 presents a flow diagram of the components of this modeling process.

Two general types of models will be used. The first type, geochemical models, estimate the leachate concentrations that result when percolating water contacts a soil or waste matrix containing contaminants. Geochemical modeling will not be used to estimate leachate derived from the waste matrix if in situ leachate or laboratory leach-test data are available (see Section 6.1.1.3). The second type, fate and transport models, predict the long-term migration potential of waste constituents after they leave the source of contamination. Together, these models produce a representation of a groundwater system that simulates transport in the groundwater system at the FEMP.

6.1.1 Geochemical Modeling

The principal objective of geochemical modeling is to estimate the concentrations of contaminants in leachate crossing the boundary between the unsaturated zone and regional aquifer. This requires the performance of a geochemical analysis, using site-specific data on in situ leachate concentrations, laboratory leach-test and TCLP data, and chemical characterization data on the waste.

6.1.1.1 Geochemical Computer Codes

Geochemical modeling will be conducted with the EQ3NR and EQ6 codes (Wolery 1983; 1984), which are industry-standard geochemical codes used to perform solubility, speciation and reaction-path calculations. Solubility and speciation calculations reveal, respectively, the maximum concentration a contaminant can have in solution and the aqueous form(s) of that contaminant

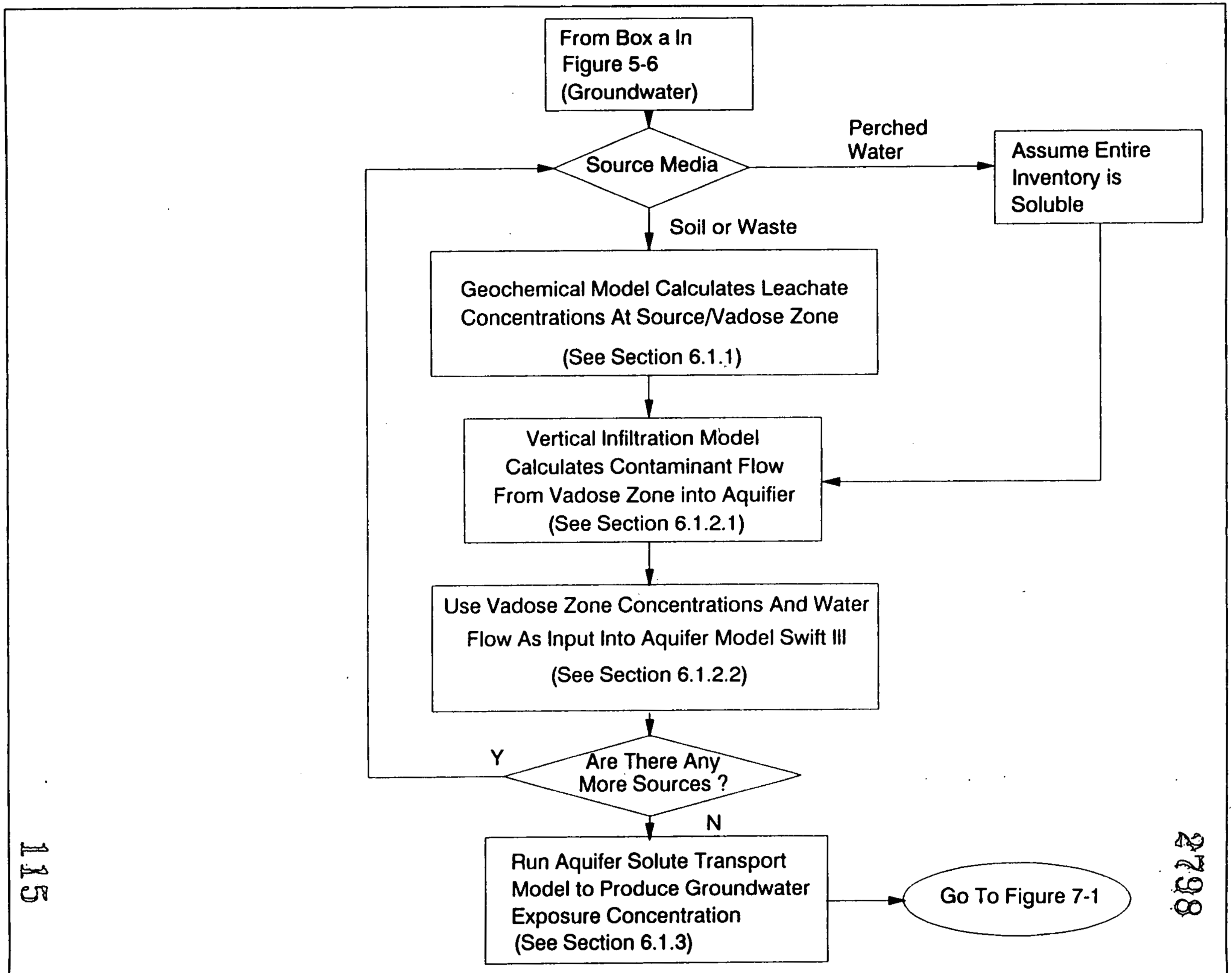

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Figure 6-1 MODELING EXPOSURE CONCENTRATION FOR GROUNDWATER

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for a specific solid/liquid/gas system. Reaction-path calculations enable a solution to migrate through, and equilibrate with, different solids. This simulates groundwater movement through compositionally distinct stratigraphic horizons.

The EQ3/6 package was developed at Lawrence Livermore National Laboratory for predicting the behavior of metals, radionuclides, and other contaminants in the natural environment. The code accesses a data base containing the thermodynamic properties of 78 elements, 862 aqueous species, 886 minerals, and 76 gases. This database includes 49 uranium-bearing aqueous species and 53 uranium-bearing minerals, constituting the most complete database available for modeling the behavior of uranium in natural waters. It also includes aqueous species and minerals of other radioactive metals (i.e., radium, thorium, etc). Total concentrations of these radioactive metals will be converted to isotopic concentrations, based on the proportion of individual isotopes present at the waste site. EQ3/6 has been validated using standard geochemistry problems, such as the speciation of sea water (Nordstrom 1979), basalt/sea water interactions (Bowers et al. 1985) and numerous comparisons with experimentally determined mineral solubilities (Jackson 1988). Benchmark comparisons were made with the results of similar codes such as PHREEQE (INTERA 1983), Nordstrom (1979), Kincaid and Morey (1984) and Kerrisk (1981).

6.1.1.2 Conceptual Geochemical Model

Prior to conducting the geochemical modeling, a conceptual model will be developed for each type of source to clarify the physical configuration simulated by the numerical model. For inorganic compounds, infiltrating rainwater reacts with the minerals in the solid waste to form a leachate within the waste unit. This is referred to as Leachate A. Leachate A migrates through the underlying glacial overburden and reacts with minerals in the glacial overburden to form Leachate B. Leachate B is assumed to reach the aquifer. Reactions referred to in the conceptual model are limited by the numerical simulation of dissolution and precipitation of mineral phases. For organic compounds, leachate concentrations will be developed using data obtained from the TCLP or by applying the EPA 70-year rule (EPA 1988a) to the inventory of organic wastes.

6.1.1.3 Geochemical Analysis

The geochemical analysis will begin by evaluating the composition of Leachate A. Figure 6-2 is a decision hierarchy that summarizes the approach to estimate Leachate A. Moving downward through this hierarchy corresponds to an increase in uncertainty and the number of assumptions required to estimate Leachate A. The least amount of uncertainty in estimating Leachate A is associated with using data obtained by the analysis of in situ leachate (e.g., leachate samples

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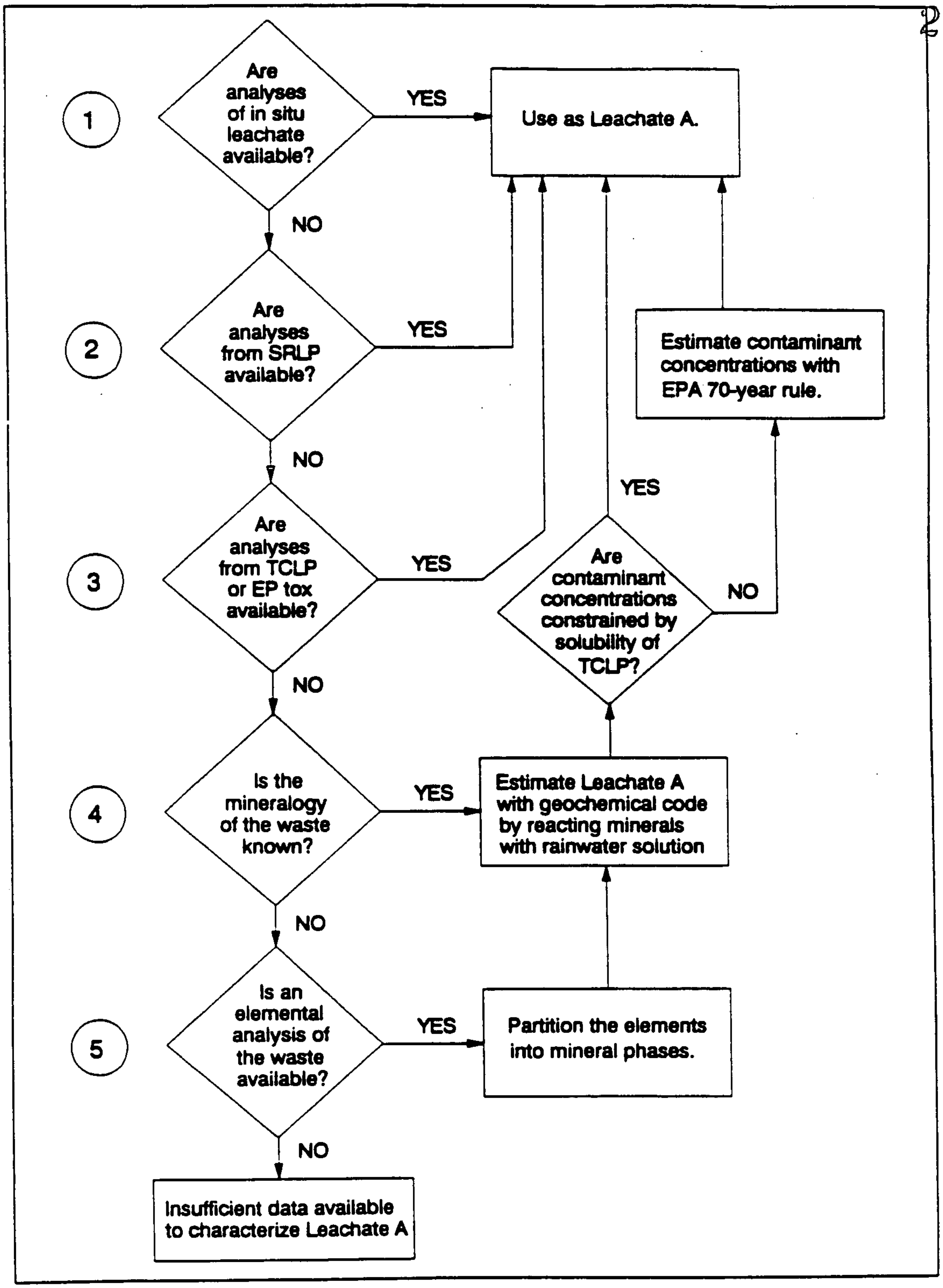

FIGURE 6-2
FLOW CHART FOR ESTIMATING LEACHATE A

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obtained from Operable Unit 1 waste pits). When in situ leachate is not available from a waste unit (e.g., waste units comprising Operable Unit 2), the waste may be analyzed by the Simulated Rainwater Leaching Procedure (SRLP) to obtain an estimate of Leachate A.

TCLP data will be available from all operable units, but these data are limited to toxic metals and organics and they do not provide a complete chemical description (e.g., anion concentrations) of Leachate A. Mineralogic data on the waste are available for some major constituents in the Operable Unit 4 silos (Litz 1974, Dettore et al. 1981 and Gill 1988) and Operable Unit 1 waste pits (NLO 1980), and these data can be used for solubility calculations. However, there are no plans to collect additional mineralogic data on the waste because information on the composition of Leachate A can be obtained in a more cost- and time- effective manner by leaching the waste and analyzing the recovered leachate. Finally, all waste units have been resampled for the purpose of further waste characterization, and as these elemental analyses become available they will be compared to previous studies (Grumski 1987, DOE 1988b, Vitro 1952 and Weston 1987) to determine if geochemical modeling needs to be repeated using the new characterization data.

When geochemical modeling is required to estimate Leachate A (an option that is executed only if leachate data are not available from source 1 or 2 in Figure 6-2), the waste minerals will be assumed to enter percolating rainwater at rates proportional to their molar abundance. This simplified approach is required because kinetic data on mineral dissolution rates are not available for the waste phases of interest. Waste that lacks mineralogic characterization can still be modeled by using the elemental analysis of the waste to partition elements into waste phases. Metals and radionuclides are combined with reported ligands (e.g., HCO_3^-, SO_4^{-2}, etc.) to form a hypothetical mineral that is known to be thermodynamically stable at the observed temperatures and pressure. For example, barium is combined with sulfate to form the mineral barite ($BaSO_4$).

After all mineral phases are determined, concentrations will be converted to moles and then partitioned into the appropriate phase (e.g., 15 ppm barium (Ba) = 1.1 E-4 moles Ba = 1.1 E-4 moles barite [$BaSO_4$]). A list identifying the contaminants of interest will be used to determine the number of waste minerals that will be modeled.

The relative proportions of each mineral in the source is then determined by dividing the moles of each mineral by the moles of the most abundant mineral in the source. These ratios will be used to calculate the relative rate that a given mineral dissolves and enters solution. As solution concentrations increase, solubility limits are reached and solid phases precipitate from the solution. When the system reaches equilibrium, the modeling is stopped. The solution composition at the termination of modeling is assumed to represent Leachate A, and this 118 composition may include silver, arsenic, barium, chromium, mercury, lead, and selenium concentrations obtained from TCLP tests. Concentrations of contaminants in Leachate A are then evaluated to determine if all contaminants are constrained by a solubility limit or TCLP value. Contaminant concentrations not constrained by either of these conditions must be reevaluated using the EPA 70-year rule (EPA 1988a). Using the 70-year rule, the concentrations of highly soluble contaminants (e.g., Cs-137 and Sr-90) are calculated by dividing 1/70 of the total inventory of each contaminant of concern by the volume of water passing through the waste in one year. Therefore, when Leachate A (Figure 6-2) is estimated with the geochemical model, contaminant concentrations will be constrained by TCLP data, solubility limits, and the EPA 70-year rule.

After the contaminant concentrations in Leachate A have been characterized, the geochemical model will be used to react Leachate A with minerals present in the glacial overburden. This reaction-path step allows for possible reduction of contaminant concentrations as the pH and composition of Leachate A is modified by reactions with silicate and carbonate minerals. Glacial overburden at the FEMP site is comprised of dolomite, quartz, feldspar, mica, clay minerals (chlorite, mica, and smectite), calcite, biotite, hornblende, and pyroxene (Solebello 1991).

The reaction-path step is simulated with the geochemical model by adding minerals in the glacial overburden to Leachate A at rates proportional to their molar abundance. The composition of Leachate A is modified by the dissolution of minerals in the glacial overburden and precipitation of both initial (i.e., glacial overburden) and secondary mineral phases. These secondary mineral phases represent minerals that are stable in the presence of leachate and glacial overburden, but they are not present in the glacial overburden prior to the introduction of leachate. When the system reaches equilibrium, the modeling is stopped. The modified leachate composition at the termination of modeling is assumed to represent the leachate composition in the glacial overburden and is referred to as Leachate B (Figure 6-2). Leachate B represents a solution that has equilibrated with minerals in the glacial overburden with respect to mineral dissolution and precipitation but not adsorption or ion exchange. Adsorption ratios estimated for glacial overburden are used in the fate and transport model to further reduce the contaminant concentrations in Leachate B prior to the leachate entering the regional aquifer. Note that the geochemical and hydrologic models are not coupled, and the geochemical processes of dissolution/precipitation and adsorption are evaluated independently.

6.1.1.4 Leaching of Organic Compounds

Concentrations of organic compounds in Leachate A will be determined using the results of TCLP tests or the 70-year rule. Organic concentrations constrained by TCLP results will be deducted from the total quantity of the contaminant in the waste at each time step simulated in the fate and transport model until the contaminant supply is exhausted. Unlike organic concentrations constrained by the 70-year rule, contaminants that have their TCLP concentration removed at each time step may persist in the waste for periods of less than or greater than 70 years. Note that a 70-year-rule concentration for a specific contaminant is based on the removal of its entire inventory over a period of 70 years. The 70-year rule is the most conservative assumption that can be made for chronic exposures since the entire contents of the waste area are assumed to be leached from the waste area in a period of 70 years (a lifetime exposure duration).

6.1.1.5 Limitations and Uncertainties of Geochemical Modeling

The geochemical analysis used to estimate leachate compositions has the following limitations:

- Only inorganic systems can be modeled with the EQ 3/6 code, and this can lead to low estimates of leachate concentrations for some constituents if organic complexation is significant.

- Adsorption and desorption (including ion exchange) processes are not considered in the EQ 3/6 calculations, yielding higher concentrations in groundwater for those contaminants that are known to sorb appreciably.

- Dissolution and precipitation kinetics must be taken as instantaneous because of insufficient kinetic data on most minerals, and this can lead to overestimation or underestimation of contaminant concentrations in groundwater.

- Mineral phases in the waste must be assumed based on the chemical composition of the waste because mineralogical data are lacking for most waste units.

- Contaminant concentrations in Leachate A that are derived with the aid of geochemical modeling are constrained by TCLP data, calculated solubility limits, and the EPA 70-year rule.

These limitations produce various degrees of uncertainty in the geochemical analysis, but only adsorption/desorption, mineralogy of the waste, and 70-year rule concentrations can be addressed on a timely basis. To this end, additional studies are in progress to evaluate the adsorption of contaminants on FEMP soils and to characterize the composition of in situ leachate. Limitations associated with thermodynamic and kinetic data require years of research to obtain critical thermodynamic data on organic phases and kinetic data on dissolution/precipitation reactions.

The uncertainties in estimating leachate compositions with this approach cannot be quantified with the available data, but the greatest uncertainties are associated with:

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- Estimating the mineralogy of the waste with the chemical analysis of the waste
- Assuming instantaneous kinetics for all dissolution and precipitation reactions
- The inability to model the thermodynamic behavior of organic compounds in the waste and adsorption processes in the glacial overburden
- Applying the 70-year rule to contaminants which do not reach solubility limits

Using in situ leachate or leachate derived from the SRLP (e.g., Operable Unit 1 and Operable Unit 2) will eliminate the uncertainty associated with bullets one and four. Uncertainty associated with adsorption processes in glacial overburden (last part of bullet three) is being addressed for leachate that has the characteristic of high pH, and these studies may be applicable to several waste units in Operable Unit 1 and Operable Unit 2.

6.1.2 Groundwater Transport Modeling

Groundwater transport models predict the long-term migration potential of waste constituents after they leave the source of contamination. At the FEMP, it is known that movement of leachate from contaminant sources to a hypothetical receptor involves flow through both an unsaturated zone (vadose zone) and saturated zone (regional aquifer and perched zones). Figure 6-3 schematically displays this vertical transport down through the unsaturated soil to the aquifer and the horizontal transport through the aquifer to the well of a potential receptor.

Vertical and horizontal migration are characterized by the bulk movement of water through the underlying geological strata. As contaminated leachate percolates from the source of contamination through the vadose zone and aquifer, its continued movement is dependent on both the physical and chemical characteristics of these formations. Predicted contaminant concentrations in groundwater will then be used in the water-dependent intake and exposure model equations presented in Section 7.0.

6.1.2.1 Transport in the Vadose Zone

This phase of contaminant transport includes the bulk migration of water and dissolved materials from source areas at the FEMP to the regional aquifer. This occurs as surface water percolates from the surface, through the source of contamination and its surrounding soil, and into the saturated zone. Downward movement of water, driven by gravitational potential, capillary potential, and other components of the total fluid potential, is the prime mover of contaminant migration through the vadose zone.

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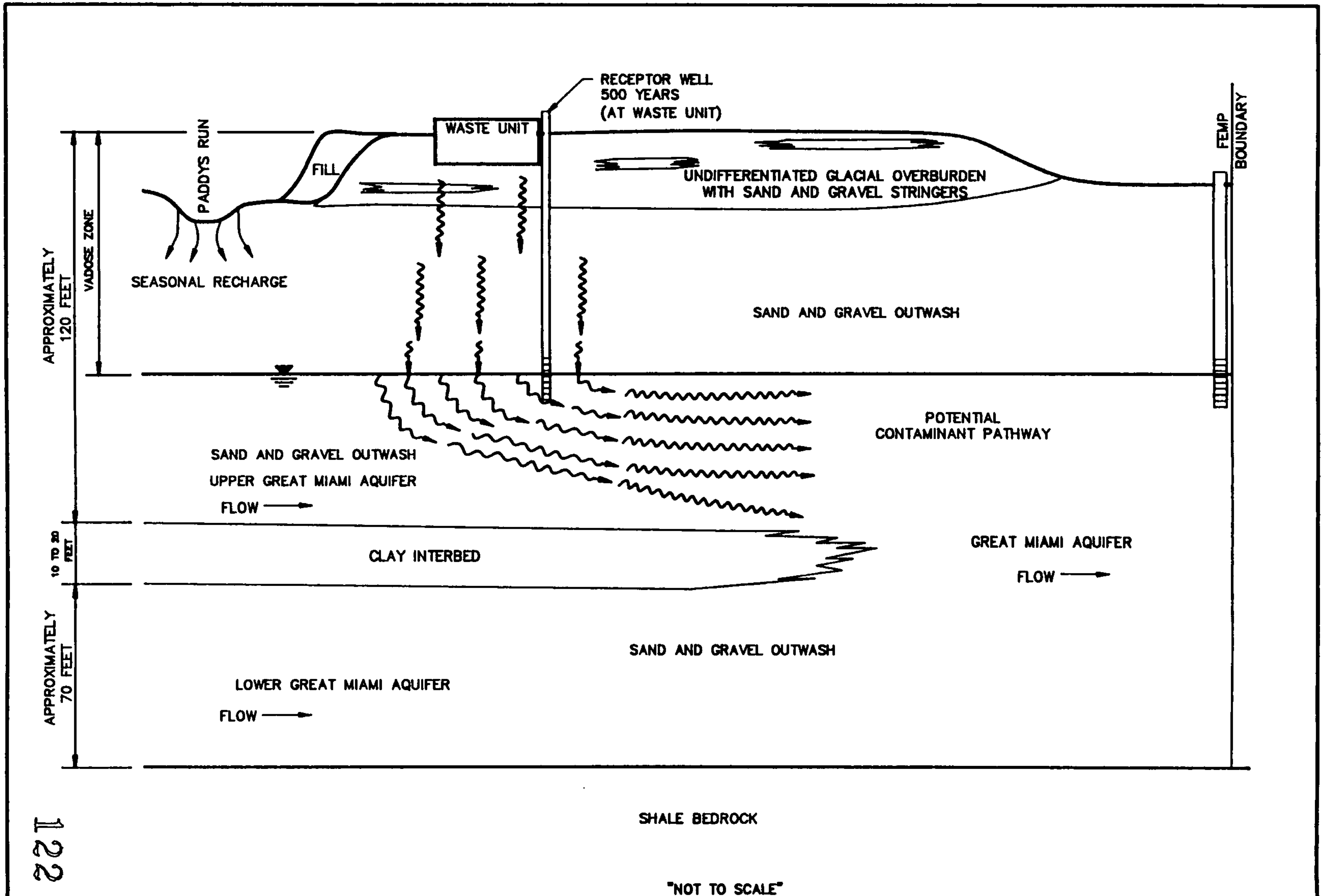

FIGURE 6-3
CONCEPTUAL FLOW MODEL

The initial concentrations will be developed using leachate data where available, and geochemical modeling for other constituents of concern (See Section 6.1.1). Each layer in the conceptual flow system will be analyzed separately, with the concentrations from the upper layers acting as the input concentrations to the lower layers. The models will assume flow is vertical through unsaturated zones. Where flowlines can be determined in the perched water zone, the one-dimensional solute transport modeling will follow the flowlines rather than following a vertical path. The one-dimensional models that will be used to simulate contaminant movement through the vadose zone will tend to produce very conservative results because they neglect transverse dispersion. The depletion of the waste source over time and radioactive decay will be taken into account in the vadose zone modeling.

6.1.2.2 Modeling Approach

The modeling approach involves completing a series of steps to develop the constituent concentrations and the mass loading at the interface of the vadose zone and the aquifer. These steps include:

- Development of a conceptual flow model based on the results of the RI/FS field investigation program
- Selection of a mathematical model to represent the conceptual model
- Use of the results of the geochemical modeling as input to the vadose zone modeling.

6.1.2.3 Vadose Zone Models

Vadose modeling is needed to provide an estimate of risk associated with contaminants that are contained in the glacial overburden and its soils. The overburden may have great capacity for immobilization and retardation of contaminants due to adsorption, precipitation, and radioactive decay. This capacity to prevent or slow the movement of contaminants to the aquifer should be evaluated with respect to future risk. The future risk posed by all potential source sites on the overburden cannot be adequately evaluated based on the fact that contamination is known to exist in the saturated portion of the Great Miami Aquifer for the following reasons:

- Relatively little of the existing contamination in the aquifer has passed through thick overburden, perhaps none.
- The degree of immobilization and retardation in thin overburden cannot be adequately estimated without vadose zone modeling. Accurate information is not available on the time and amount of contaminant introduction to the overburden, consequently, useful direct determinations cannot be made.

- Some contaminant species present in the vadose zone may not have reached the water table in the Great Miami Aquifer.

Analytical models were selected for use, based upon the following factors:

- Analytical methods are the most efficient alternative when data necessary for the characterization of the system is sparse and uncertain. At the FEMP, data pertaining to the unsaturated zone and many of the constituents of concern are generally lacking.

- The method is consistent with approaches used for similar radionuclide assessment codes such as the flow portions of PRESTO (EPA 1987b) and other site studies.

- The basis of the solution is well documented and the code has been extensively verified.

The following criteria were used in selecting specific analytical models:

- Capability of treating adsorption, radioactive decay, and longitudinal dispersion
- Capability of calculating concentrations at large times and distances
- Availability of code
- Degree of code documentation
- Degree of code verification

The models selected to evaluate flow in the vadose zone are ST1D (IT 1990), and ODAST (Javendel et al. 1984). ST1D, a one-dimensional analytical solution, will be used for the initial screening of constituents for mobility. ODAST, also a one-dimensional analytical solution, will be used for determining fate and transport of the remaining constituents in the unsaturated zone. These computer codes are based on the solution originally developed by Ogata and Banks (1961), and calculate the normalized concentrations of a given constituent in a uniform flow field from a source having a constant or varying concentration in the initial layer. The ODAST code can account for retardation of contaminants, source changes, and decay. ST1D and ODAST have been extensively verified against STRIP1B (Batu 1989). The use of other analytical models for transport in the vadose zone is not anticipated. However, if a case is discovered, where simple analytical models cannot be used, a more detailed model such as SESOIL (EPA 1984a), which simulates volatilization, hydrolysis and complexation, may be substituted. Any other model used to satisfy special needs will be subject to EPA approval. STRIP1B may be used to cross-check results obtained from ST1D and ODAST.

Vadose zone models will be checked for consistency with historic concentration data to the extent possible. If historic concentrations are available in or near the contaminant pathway being modeled, and if any information is available on historic source loading, then the model will be run using the source loading information to see if the calculated concentrations approximate the measured concentrations. If the calculated concentrations do not approximate the historic concentrations, appropriate parameters will be adjusted to produce a model that is adequate for risk assessment.

6.1.3 Transport in the Aquifer

This phase of contaminant transport involves the advective and diffusive migration of water and dissolved materials from one part of the Great Miami Aquifer to another. As contaminated leachate percolates from the vadose zone into the saturated zone of the aquifer, its continued movement is dependent on physical and chemical characteristics of the aquifer (Figure 6-3). The physical properties of the aquifer influence the bulk movement of water, and the chemical and physical properties influence the ease with which the aquifer allows the migration of specific contaminants.

6.1.3.1 Great Miami Aquifer Model

The groundwater flow and solute transport model contained in the Sandia Waste Isolation Flow and Transport (SWIFT III) computer code (Geotrans 1987) will be used to analyze contaminant transport in the regional aquifer. The SWIFT III code is a fully transient three-dimensional, finite-difference model which solves coupled equations describing water flow and transport in geologic media. The SWIFT III program consists of a main routine and about 70 supporting subroutines.

The model, applied at the FEMP since 1988, has been extensively calibrated against known uranium concentrations in groundwater. The SWIFT III code and its verification and application are fully outlined in the Flow and Solute Transport Computer Code Verification Report (IT 1990), along with the input parameters used. Even though other constituents were not considered in the calibration, this does not change the flow model and the model can be applied to other contaminants. The magnitude of uncertainty for other contaminants will depend on the uncertainty in the projection of attenuation and retardation of the contaminants.

6.1.4 Parameter Selection

Quantification of phenomena affecting water movement and contaminant transport is one of the major concerns during any effort to model groundwater flow at the FEMP. Table 6-2 represents typical values for parameters at the FEMP. Some parameters for the aquifer shown in Table 6-2

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TABLE 6-2
REPRESENTATIVE FLOW PARAMETERS FOR THE FEMP[a]

Parameter	Vadose Zone	Aquifer[b,c]
Porosity (%)	22 - 39	25
Specific Yield (%)	6 - 25	---
Bulk Density (g/cm^3)	1.6 - 1.8	1.7 - 2.0
Field Capacity (%)	14 - 28	---
Dispersion coefficient		
Longitudinal (cm^2/sec)	7.63E-6 - 2.50E-3	1.17 - 10.67
Transverse (cm^2/sec)	---	0.117 - 1.07
Hydraulic Conductivity		
Horizontal (cm/sec)	2.50E-6 - 0.16	0.16 - 0.212
Vertical (cm/sec)	1.25E-7 - 0.016	0.016 - 0.021
Seepage Velocity		
Horizontal (cm/sec)	---	3.85E-4 - 3.50E-3
Vertical (cm/sec)	3.52E-7 - 9.17E-6	---

[a] RI/FS Database
[b] Values obtained from SWIFT III calibration
[c] Values representative only for the sand and gravel aquifer and not for the clay interbed that is present beneath the site dividing the aquifer

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represent the mean values obtained from the calibration of the groundwater model. Hydraulic conductivity and porosity for the aquifer are also included in the ranges for the vadose zone because the upper part of the aquifer is not saturated and is part of the vadose zone. Parameters applied to vadose modeling will be subject to continued investigation as the vadose modeling progresses. The continued investigation will include continued search of pertinent scientific literature, geochemical investigations related to partition coefficients, and checks for consistency between model results and historic data.

Uncertainty in the selection of model parameter values will be addressed by performing sensitivity analyses. Sensitivity analyses will be performed by varying parameters within reasonable ranges. Analyses will yield a range of predicted exposure point concentrations that may be used in risk assessments.

6.1.4.1 Moisture Content

The moisture content is the amount of moisture held within the vadose zone at any given time. This moisture content, or degree of saturation, will vary continuously over time and along flow paths. It directly affects the ability of a material to pass fluids (hydraulic conductivity) and the capillary effects keeping water within the material. This moisture content can vary from saturation to air dryness (Hillel 1982).

Site specific information will be used where available. Where the moisture content of the vadose zone is not available, the moisture content will be estimated by one or two methods. The first technique is based upon Clapp and Hornberger's equation (1978) as presented in the Exposure Assessment Manual (EPA 1988c). This equation states that:

$$\Theta = (\Theta_S)(q/K_S)^{1/(2b+3)} \qquad (6\text{-}1)$$

where

Θ	=	Moisture content in the vadose zone (unitless)
Θ_S	=	Saturated moisture content in the vadose zone (unitless)
q	=	Infiltration or recharge rate (m/s)
K_S	=	Saturated vertical hydraulic conductivity (m/s)
b	=	Soil specific exponential parameter (unitless)
1/(2b+3)	=	Soil specific exponential parameter factor estimated from EPA (1987a)

The second technique is based upon the relationship:

$$r = n - Sy \qquad (6\text{-}2)$$

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where

r = Specific retention or minimum moisture content (unitless)
n = Porosity (unitless)
Sy = Specific yield (unitless)

6.1.4.2 <u>Hydraulic Conductivity</u>

The most important difference between unsaturated and saturated flow is hydraulic conductivity. When the matrix is saturated, all of the pores are water-filled and conducting, so that conductivity is at its maximum. When the matrix dries, some of the pores fill with air and the conductive portion of the unconsolidated material decreases. The first pores to drain are the larger more conductive ones, leaving only the smaller, less conductive pores available for water movement. Furthermore, as the water drains, increasing capillary forces trap water in matrix pores.

The unsaturated hydraulic conductivity is commonly estimated based on a relationship between the soil moisture curve and saturated hydraulic conductivity using techniques such as those found in van Genuchten (1978). However, at the FEMP no measurements of water content, matric suction, or unsaturated hydraulic conductivity have been completed. Therefore, it will be necessary to rely on estimates, and where available, direct measurements of saturated hydraulic conductivity for the hydraulic conductivity of the vadose zone. Typical hydraulic conductivities for the vadose zone at the FEMP are listed in Table 6-2. When these estimates are applied to the calculation of velocity, they will be adjusted to reflect partial saturation.

The use of saturated hydraulic conductivities will tend to overestimate the movement of fluids through the vadose zone. However, given the long period of time for this analysis (up to 1000 years), this overestimation will not have a major impact on the analysis.

6.1.4.3 <u>Specific Yield</u>

The specific yield is a measure of the amount of water that is released from storage as the water level in an aquifer declines. For the purposes of this analysis, the specific yield will be used to estimate the moisture content of the vadose zone material. Estimates for the specific yield will be obtained from RI/FS sampling, or derived from published tables found in Morris and Johnson (1967), and van der Leeden et al. (1990).

6.1.4.4 <u>Porosity</u>

The porosity of a material is a measure of the voids or pore space within a material as compared to the total volume. Porosity is important in determining the velocity of fluids in saturated zones and in estimating values for the moisture content. Measured porosities at the FEMP will be

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obtained from site RI/FS samples. Additional data may be obtained from porosities listed in published tables found in Morris and Johnson (1967), Driscoll (1986), and van der Leeden et al. (1990).

6.1.4.5 Vertical Seepage Velocity

The estimates of the flow parameters were used to calculate the seepage velocity for input into the vadose zone transport model. To determine whether flow was occurring as a saturated front, infiltration (q) was compared to the vertical hydraulic conductivity (K_v). If $q \geq K_v$, it is assumed that saturated conditions exist and velocity is calculated based upon the following formula:

$$V_{pw} = (K_v)(i)/n \qquad (6\text{-}3)$$

where

V_{pw} = Seepage velocity (m/s)
K_v = Vertical hydraulic conductivity (m/s)
i = Hydraulic gradient (m/m)
n = Porosity (unitless)

If $q < K_v$, it will be assumed that a seepage would not occur under saturated conditions and the following formula would then be used to calculate the seepage velocity:

$$V_{pw} = q/\Theta \qquad (6\text{-}4)$$

where

q = Infiltration (m/s)
Θ = Moisture content (unitless)

Based on the assumptions of steady-state moisture content, the selected K value, and using the best field data available for hydraulic gradient, the calculated seepage velocity will be conservative and tend to overestimate the rate of fluid movement.

6.1.4.6 Partition Coefficients

As contaminated leachate flows through a geologic formation, the individual contaminants may react with the solids in the formation in a variety of degrees and ways. This slows the transport of these contaminants. Partition coefficients, or "K_d's", are used to account for this phenomenon in the transport equation. A contaminant's K_d expresses the ratio of its concentration in the solid and liquid components in the groundwater flow system, at a given location in that system. The

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use of K_d values assumes that a linear equilibrium relationship exists between the solid and solution phase concentrations of a contaminant.

Site-specific K_d values are currently available only for some mobile uranium compounds at the site. A literature search will be completed to determine appropriate K_d values for the remaining inorganic and radioactive constituents. Values found in the literature search will be carefully screened to select those values that will be derived under conditions that approximate those at the FEMP.

When parameter values derived from literature are used, it is imperative that K_d values from similar environments be considered. Similar soil types and water compositions should be used to generate the values. Criteria used to determine similarities in soil types include: pH, E_H, mean arithmetic particle diameter, total organic carbon (TOC), cation exchange capacity (CEC), and free ion oxides (FIO). This may prove difficult in terms of matching groundwater compositions because most studies use dilute acid solutions spiked with the compound of interest and do not represent natural conditions. However, these studies can provide an initial estimate of interaction between the contaminant and the solid matrix. The use of literature K_d values may result in retardation values that differ from site-specific conditions, and would result in uncertainty in the estimate of contaminant concentration at the receptor.

When a site-specific or literature-based K_d value is not available for a given organic chemical, its K_d value can be calculated, using an organic carbon partitioning coefficient, or "K_{oc}", the amount of carbon present in the soil matrix, and the size distribution of the matrix material in the vadose zone (Mills et al. 1985):

$$K_d = K_{oc}\,[0.2(1-f)x^s_{oc} + (f)(x^f_{oc})] \qquad (6\text{-}5)$$

where

K_d	=	Soil partitioning coefficient (mL/g)
K_{oc}	=	Organic carbon partitioning coefficient (mL/g)
f	=	Mass fraction of silt or clay (unitless)
x^s_{oc}	=	Organic carbon content of sand (unitless)
x^f_{oc}	=	Organic carbon content of silt-clay (unitless)

The numerical values for (f), (x^s_{oc}), and (x^f_{oc}) will be site-specific. The K_{oc} is the partition coefficient of a contaminant between water and a 100% organic carbon representing the organic material present in soil or sediment. Chemical-specific values for K_{oc} are available in the literature for many organic compounds. Additional K_{oc} values may be calculated using empirical

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formulas relating the octanol-water partitioning coefficient (K_{ow}) to the K_{oc}. The K_{ow} (mL/mL) is the ratio of a contaminant's concentrations in a system containing water and octanol. The K_{ow}'s for the remaining constituents of concern are found in the U.S. EPA Water Engineering Research Laboratory Treatability Database, Cincinnati, Ohio supplemented by Howard (1990), Montgomery (1990 and 1991), and Verschueren (1983), if necessary. The formula (Mills et al. 1985) used to relate K_{ow} to K_{oc} is:

$$K_{oc} = (0.63)(K_{ow}) \qquad (6\text{-}6)$$

This approach of using empirical relationships assumes K_d is problem-specific because it depends on both the chemical modeled and the soil type, while K_{oc} is a property only of the chemical modeled. [While this approach is generally acceptable, Cleary et al. (1991) present laboratory evidence for five volatile organic compounds in eight different soils which shows K_{oc} is not a fixed property of the chemical in question.] Their study raises questions on the standard use of K_{oc} values. However, the standard approach given by Equation 6-6 appears to be reasonable given the lack of site-specific data.

Estimated K_d values for the major contaminants at the FEMP have been determined and are presented in Tables 6-3 and 6-4. Chemical forms of these radionuclides and metals generally have significant effects on partitioning coefficients and will be evaluated along with site-specific analytical data. Radioactive decay products (progeny) of the radionuclides at the FEMP may not have the same partitioning coefficients as the parent. The impact of such differences on fate and transport modeling results will be evaluated. These estimates of K_d values are acceptable for screening purposes, and conservative transport assessment.

The partitioning coefficient may also used to derive a retardation factor (Rf). Though the K_d/Rf formulation of the reaction term of the transport equation has numerous assumptions and uncertainties associated with it, it nevertheless provides a practical means of incorporating the reaction process into transport models.

6.1.4.7 Effects of Radioactive Decay and Biodegradation

Nuclear, chemical, and biological processes play major roles in the fate of some contaminants, and are thus an important aspect of all fate and transport modeling. For example, concentrations of both radioactive isotopes and organic compounds remaining in the environment for long periods would be greatly overestimated without accounting for the effects of radioactive decay and biodegradation. Therefore, information about radioactive decay and environmental degradation is used in several of the transport models.

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TABLE 6-3
PARTITIONING COEFFICIENTS FOR RADIONUCLIDES AND INORGANICS AT THE FEMP[a]

Constituents	Vadose Layer 1 (Clay)		Vadose Layer 2 (Sand & Gravel)	
	K_d (ml/g)	Reference	K_d (ml/g)	Reference
Radionuclides				
Ac-227	2.40E+03	Sheppard & Thibault 1990	4.50E+02	Sheppard & Thibault 1990
Cs-137	1.81E+03	Sheppard et. al. 1984	1.37E+03	Sheppard et. al. 1984
Np-237	5.50E+01	Sheppard & Thibault 1990	5.00E+00	Sheppard & Thibault 1990
Pa-231	2.70E+03	Sheppard & Thibault 1990	5.50E+02	Sheppard & Thibault 1990
Pb-210	3.00E+03	Gerritse et. al. 1982	3.80E+01	Raj and Zachara 1984
Pu-238	1.70E+03	Glover et. al. 1976	1.00E+02	Glover et. al. 1976
Pu-239/240	1.70E+03	Glover et. al. 1976	1.00E+02	Glover et. al. 1976
Ra-224	6.96E+02	Gillham et. al. 1981	1.06E+02	Sheppard et. al. 1984
Ra-226	6.96E+02	Gillham et. al. 1981	1.06E+02	Sheppard et. al. 1984
Ra-228	6.96E+02	Gillham et. al. 1981	1.06E+02	Sheppard et. al. 1984
Ru-106	8.00E+02	Sheppard & Thibault 1990	5.50E+01	Sheppard & Thibault 1990
Sr-90	1.00E+01	Sheppard et. al. 1984	2.50E+00	Sheppard et. al. 1984
Tc-99	1.18E-01	Sheppard et. al. 1984	7.00E-02	Sheppard et. al. 1984
Th-228	5.80E+03	Sheppard & Thibault 1990	3.20E+03	Sheppard & Thibault 1990
Th-230	5.80E+03	Sheppard & Thibault 1990	3.20E+03	Sheppard & Thibault 1990
Th-232	5.80E+03	Sheppard & Thibault 1990	3.20E+03	Sheppard & Thibault 1990
U-234	1.80E+00	DOE 1989	1.48E+00	DOE 1989
U-235/236	1.80E+00	DOE 1989	1.48E+00	DOE 1989
U-238	1.80E+00	DOE 1989	1.48E+00	DOE 1989

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TABLE 6-3
(Continued)

Constituents	Vadose Layer 1 (Clay)		Vadose Layer 2 (Sand and Gravel)	
	Kd (ml/g)	Reference	Kd (ml/g)	Reference
Inorganics				
Aluminum	1.50E+03	Baes and Sharp 1984	1.5E+03	Baes and Sharp 1984
Arsenic	2.00E+02	Baes and Sharp 1984	2.00E+02	Baes and Sharp 1984
Antimony	2.50E+02	Sheppard & Thibault 1990	4.50E+01	Sheppard & Thibault 1990
Barium	1.14E+03	Gillham et. al. 1981	2.00E+01	Sheppard et. al. 1984
Beryllium	1.30E+03	Sheppard & Thibault 1990	2.50E+02	Sheppard and Thibault 1990
Cadmium	5.00E+02	Gerritse et. al. 1982	1.20E+01	Raj and Zachara 1984
Calcium	5.00E+01	Sheppard and Thibault 1990	5.00E+00	Sheppard and Thibault 1990
Chromium	1.50E+03	Sheppard and Thibault 1990	7.00E+01	Sheppard and Thibault 1990
Cobalt	5.50E+02	Sheppard and Thibault 1990	6.00E+01	Sheppard and Thibault 1990
Copper	1.25E+02	Gerritse et. al. 1982	3.50E+01	Baes and Sharp 1984
Iron	1.65E+02	Sheppard and Thibault 1990	2.20E+02	Sheppard and Thibault 1990
Lead	3.00E+03	Gerritse et. al. 1982	3.80E+01	Raj and Zachara 1984
Magnesium	4.50E+00	Baes and Sharp 1984	4.50E+00	Baes and Sharp 1984
Manganese	1.80E+02	Sheppard and Thibault 1990	5.0E+01	Sheppard and Thibault 1990
Mercury	1.00E+01	Baes and Sharp 1984	1.00E+01	Baes and Sharp 1984
Molybdenum	9.00E+01	Sheppard and Thibault 1990	1.0E+01	Sheppard and Thibault 1990
Nickel	6.50E+02	Sheppard and Thibault 1990	4.00E+02	Sheppard and Thibault 1990
Potassium	7.50E+01	Sheppard and Thibault 1990	1.50E+01	Sheppard and Thibault 1990
Selenium	7.40E+02	Sheppard and Thibault 1990	1.50E+02	Sheppard and Thibault 1990
Silver	1.80E+02	Sheppard and Thibault 1990	9.00E+01	Sheppard and Thibault 1990
Sodium	1.00E+02	Baes and Sharp 1984	1.00E+02	Baes and Sharp 1984
Thallium	1.50E+03	Baes and Sharp 1984	1.50E+03	Baes and Sharp 1984
Vanadium	1.00E+03	Baes and Sharp 1984	2.00E+02	Gerritse et. al. 1982
Zinc	2.40E+03	Sheppard and Thibault 1990	2.00E+02	Sheppard and Thibault 1990

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TABLE 6-4
PARTITIONING COEFFICIENTS FOR ORGANIC COMPOUNDS AT THE FEMP[a]

Constituents	K_{ow}[b] (mL/mL)	K_{oc} (mL/g)	K_d[c] Vadose 1 (mL/g)	K_d[c] Vadose 2 (mL/g)
1,1-Dichloroethane	6.17E+01	3.89E+01	1.18E+00	5.10E-01
1,1-Dichloroethene	3.02E+01	1.90E+01	5.78E-01	2.50E-01
1,1,2-Trichloro-1,2,2-trifluoroethane[d]	1.48E+02	9.32E+01	2.83E+00	1.22E+00
1,1,1-Trichloroethane	2.95E+02	1.86E+02	5.65E+00	2.44E+00
1,1,2,2-Tetrachloroethane	2.46E+02	1.55E+02	4.70E+00	2.03E+00
1,2-Dichloroethene[e]	1.23E+02	7.75E+01	2.36E+00	1.02E+00
2-Butanone	1.81E+00	1.14E+00	3.47E-02	1.50E-02
2-Methylnaphthalene	7.24E+03	4.56E+03	1.39E+02	5.98E+01
2-Methyl phenol	8.91E+01	5.61E+01	1.71E+00	7.36E-01
2-Propanol	6.90E-01	4.35E-01	1.32E-02	5.70E-03
2,4-Dimethyl phenol	2.63E+02	1.66E+02	5.04E+00	2.17E+00
4-Methyl-2-Pentanone[e]	1.23E+01	7.75E+00	2.36E-01	1.02E-01
4-Methyl phenol[e]	7.94E+01	5.00E+01	1.52E+00	6.57E-01
Acenaphthene	8.32E+03	5.24E+03	1.59E+02	6.88E+01
Acetone	5.70E-01	3.59E-01	1.09E-02	4.71E-03
Anthracene	2.80E+04	1.76E+04	5.36E+02	2.31E+02
Aroclor-1016	2.40E+04	1.51E+04	4.60E+02	1.98E+02
Aroclor-1242	1.29E+04	8.11E+03	2.47E+02	1.06E+02
Aroclor-1248	5.62E+05	3.54E+05	1.08E+04	4.65E+03
Aroclor-1254	1.07E+06	6.75E+05	2.05E+04	8.86E+03
Aroclor-1260	1.29E+06	8.13E+05	2.47E+04	1.07E+04
Benzene	1.35E+02	8.51E+01	2.59E+00	1.12E+00
Benzo(a)anthracene	4.00E+05	2.52E+05	7.66E+03	3.31E+03
Benzo(a)pyrene	9.55E+05	6.02E+05	1.83E+04	7.89E+03
Benzo(b)fluoranthene	3.72E+06	2.34E+06	7.11E+04	3.07E+04
Benzo(g,h,i)perylene	1.70E+07	1.07E+07	3.25E+05	1.40E+05
Benzo(k)fluoranthene	6.92E+06	4.36E+06	1.32E+05	5.72E+04

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TABLE 6-4
(Continued)

Constituents	K_{ow}[b] (mL/mL)	K_{oc} (mL/g)	K_d[c] Vadose 1 (mL/g)	K_d[c] Vadose 2 (mL/g)
Benzoic acid	7.41E+01	4.67E+01	1.42E+00	6.13E-01
Beta-BHC	6.31E+03	3.98E+03	1.21E+02	5.22E+01
Bis(2-ethylhexyl)phthalate	2.00E+05	1.26E+05	3.83E+03	1.65E+03
Butylbenzyl phthalate	6.03E+04	3.80E+04	1.15E+03	4.98E+02
Carbondisulfide	1.45E+02	9.14E+01	2.78E+00	1.20E+00
Carbon tetrachloride	5.37E+02	3.38E+02	1.03E+01	4.44E+00
Chloroform	9.33E+01	5.88E+01	1.79E+00	7.71E-01
Chlordane	6.03E+02	3.80E+02	1.15E+01	4.98E+00
Chrysene	4.00E+05	2.52E+05	7.66E+03	3.31E+03
cis-1,2-Dichloroethene[e]	3.02E+01	1.90E+01	5.78E-01	2.50E-01
Cyanide	2.24E+00	1.41E+00	4.29E-02	1.85E-02
DDT	1.55E+06	9.77E+05	2.97E+04	1.28E+04
Dibenzofuran	1.32E+04	8.32E+03	2.53E+02	1.09E+02
Dibenzo(a,h) anthracene	9.33E+05	5.88E+05	1.79E+04	7.71E+03
Di-n-butyl phthalate	1.58E+05	9.95E+04	3.03E+03	1.31E+03
Di-n-octyl phthalate	1.58E+09	9.95E+08	3.03E+07	1.31E+07
Ethylbenzene	1.40E+03	8.82E+02	2.68E+01	1.16E+01
Ethyl parathion[e]	5.75E+03	3.63E+03	1.10E+02	4.76E+01
Fluoranthene	2.14E+05	1.35E+05	4.09E+03	1.77E+03
Fluorene	1.50E+04	9.45E+03	2.87E+02	1.24E+02
Ideno(1,2,3-cd)pyrene	4.57E+07	2.88E+07	8.75E+05	3.78E+05
Methyl parathion	1.10E+02	6.93E+01	2.11E+00	9.09E-01
Methylene chloride	1.78E+01	1.12E+01	3.41E-01	1.47E-01
Naphthalene	2.30E+03	1.45E+03	4.40E+01	1.90E+01
N-Nitrosodiphenyl amine	1.35E+03	8.51E+02	2.59E+01	1.12E+01
Pentachlorphenol	1.02E+05	6.43E+04	1.95E+03	8.43E+02
Phenol	2.88E+01	1.81E+01	5.52E-01	2.38E-01
Phenanthrene	2.90E+04	1.83E+04	5.55E+02	2.40E+02

TABLE 6-4
(Continued)

Constituents	K_{ow}[b] (mL/mL)	K_{oc} (mL/g)	K_d[c] Vadose 1 (mL/g)	K_d[c] Vadose 2 (mL/g)
Pyrene	1.51E+05	9.54E+04	2.90E+03	1.25E+03
Tetrachloroethene	3.39E+02	2.13E+02	6.49E+00	2.80E+00
Toluene	4.90E+02	3.09E+02	9.38E+00	4.05E+00
Trichloroethene	3.39E+02	2.13E+02	6.49E+00	2.80E+00
Total Xylenes	1.10E+03	6.93E+02	2.11E+01	9.09E+00
Vinyl Chloride	3.98E+00	2.51E+00	7.62E-02	3.29E-02

a This table presents default values, which are subject to change, based on FEMP-site site-specific data.
b K_{ow} taken from EPA Treatability Database (1990).
c Calculated by Equation 6-5.
d K_{ow} data are not available in the EPA Treatability Database (1990). K_{ow} data from Howard (1990).
e K_{ow} data are not available in the EPA Treatability Database (1990). K_{ow} data from Montgomery et al. (1990).

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Radioactive Decay

The decrease in the quantity of a radioactive material over time is calculated by the exponential decay relationship,

$$A = A_o e^{-\lambda_{ri} t} \quad (6\text{-}7)$$

where

A =activity at time, t
A_o =activity at time, t=0
λ_{ri} =radioactive decay constant, given by:

$$\lambda_{ri} = \ln(2)/T_{1/2} \quad (6\text{-}8)$$

where

$T_{1/2}$ =radioactive half-life (yr)

Half-lives and radiological decay constants for some of the radiological constituents at the FEMP are presented in Table 6-5. Equation 6-8 will be used to calculate any additional decay coefficients which may be needed in support of fate and transport modeling at the site.

Environmental Degradation

The source used to determine degradation rates for organic chemicals in air, soil, and water is the Handbook of Environmental Degradation Rates (Howard et al. 1990), which was produced by Syracuse Research Corporation for the U.S. EPA to support the superfund Amendment and Reauthorization Act (SARA) Section 313. The major sources of degradation rates reviewed for the book were U.S. EPA data bases including CHEMFATE, BIOLOG, and BIODEG. CHEMFATE and BIODEG contain actual experimental data. Each of the organics at the site will eventually degrade at a rate that can be calculated from information on half-lives in pertinent environmental media.

Reported half-lives reflect only degradation processes, not other transport processes such as volatilization. For the most soluble organics, biotic biodegradation is the principal means of degradation in the groundwater. The abiotic process of hydrolysis is important, but to a lesser extent. Other abiotic reactions, such as photolysis and oxidation/reduction, do not play an important role in degradation. A range of half-lives is available for most chemicals in each

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TABLE 6-5
DECAY RATES (λ_{ri}) OF SELECTED RADIONUCLIDES AT THE FEMP

Nuclide	Half-Life (years)	Decay Constant (yr^{-1})	Decay Constant (d^{-1})
Ac-227	2.177E+01	3.184E-02	8.722E-05
Am-241	4.322E+02	1.604E-03	4.394E-06
Cs-137	3.017E+01	2.297E-02	6.294E-05
Np-237	2.140E+06	3.239E-07	8.874E-10
Pa-231	3.276E+04	2.116E-05	5.797E-08
Pb-210	2.226E+01	3.114E-02	8.531E-05
Pu-238	8.775E+01	7.899E-03	2.164E-05
Pu-239/240	2.413E+04	2.872E-05	7.870E-08
Ra-224	9.918E-03	6.989E+01	1.915E-01
Ra-226	1.600E+03	4.332E-04	1.187E-06
Ra-228	5.760E+00	1.203E-01	3.297E-04
Ru-106	1.009E+00	6.871E-01	1.883E-03
Sr-90	2.860E+01	2.424E-02	6.640E-05
Tc-99	2.130E+05	3.254E-06	8.916E-09
Th-228	1.913E+00	3.623E-01	9.926E-04
Th-230	7.700E+04	9.002E-06	2.466E-08
Th-232	1.405E+10	4.933E-11	1.352E-13
U-234	2.445E+05	2.835E-06	7.767E-09
U-235	7.038E+08	9.849E-10	2.698E-12
U-238	4.468E+09	1.551E-10	4.250E-13

environmental compartment, e.g., air, soil, water. In general, biodegradation rates in groundwater are slower than for soil and surface water because groundwater is often limited in terms of microbial populations. Rates are often one-half the rate in surface water.

Using the half-life ($T_{1/2}$) data, the method for determining the degradation coefficient is (Petrucci 1977):

$$\lambda_c = \ln(2)/T_{1/2} \tag{6-9}$$

and $1/\lambda_c$ produces coefficients in terms of time.

For groundwater and vadose zone modeling, the most conservative value (e.g., the smallest half-life) will be used. This is usually the factor of anaerobic biodegradation.

6.2 SURFACE WATER TRANSPORT MODELING

Figure 6-4 depicts the modeling approach that will be used to estimate contaminant concentrations in surface water and sediment resulting from transport by surface water runoff. Modeling the transport of soil by runoff requires characterization of the contaminants in the initial soil or waste source term. Once a runoff scenario is selected, one of two models will be used to quantify the migration of contaminated soil to stream sediment from erosion by surface water runoff. The two soil loss models, obtained from the EPA Superfund Exposure Assessment Manual (EPA 1988c), are the Universal Soil Loss Equation (USLE) and the Modified Universal Soil Loss Equation (MUSLE). These models calculate the total mass of soil transported each year. The USLE model takes the same form as MUSLE, except that USLE uses an area dependent method to determine runoff, while MUSLE employs event-specific runoff volume and flowrate variables.

Soil loss is estimated using the USLE:

$$Y(s)_A = [(R_r)(SA)(S_d)](K)(\Delta)(C)(Z) \tag{6-10}$$

Soil loss is estimated using the MUSLE:

$$Y(s)_E = (CF)[(V_r)(q_p)]^{0.56}(K)(\Delta)(C)(Z) \tag{6-11}$$

where

$Y(s)_A$ =Annual soil loss in runoff (metric tons/yr)
$Y(s)_E$ =Soil loss in runoff (metric tons per event)
CF =Conversion factor (11.8 for metric units)
R_r =Rainfall and runoff erosion potential factor (unitless)

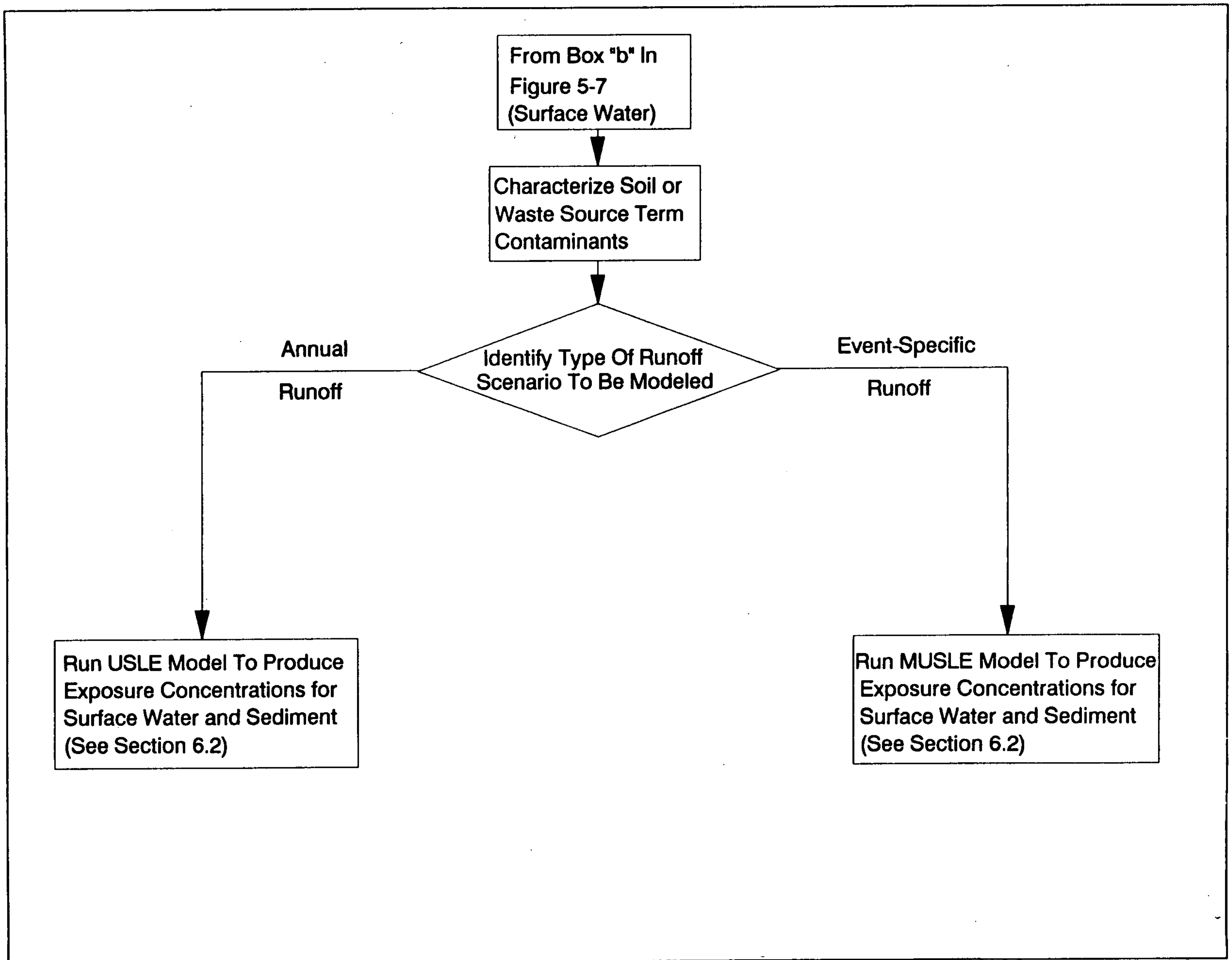

Figure 6-4 MODELING EXPOSURE CONCENTRATION FOR SURFACE WATER AND SEDIMENT VIA RUNOFF

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K = Soil erodibility factor (metric tons/ha/unit R_r)
Δ = Product of slope length factor and slope steepness factor (unitless)
C = Cover factor (unitless ratio)
Z = Erosion control practice factor (unitless)
SA = Contaminated area (hectares, ha)
S_d = Sediment delivery ratio (unitless)
D = Overland distance between site and receiving water body (ft)
V_r = Volume of runoff (m^3)
q_p = Peak runoff flow rate (m^3/sec)

Additional models are used to describe contaminant partitioning between soil and water in the receiving water body. These partitioning models provide an estimate of the contaminant concentration in surface water runoff and in the soil that is carried with the runoff and deposited in the sediments of receiving surface water bodies (Haith 1980, Mills et al. 1982). The portion of contaminant from the eroded soil that remains with the sediment or is dissolved in the water is estimated using the following equations, respectively:

$$S_s = [1/(1+O_c/(K_d \bullet \rho))]\,(C_s)(X_s) \qquad (6\text{-}12)$$

and

$$M_s = [1/(1+(K_d \bullet \rho)/O_c)]\,(C_s)(X_s) \qquad (6\text{-}13)$$

where

S_s = Absorbed quantity of contaminant (portion to sediment) (mg)
M_s = Dissolved quantity of contaminant (portion to water) (mg)
O_c = Available water capacity in top cm of soil (unitless)
K_d = Sorption partition coefficient (cm^3/g)
ρ = Bulk soil density (g/cm^3)
C_s = Concentration of contaminant in soil (mg/kg)
X_s = Soil loss in runoff (kg)

The default value for O_c at the site is 0.6 and the contaminant concentration in sediment of the receiving water body is:

$$C_s = S_s/X_s \qquad (6\text{-}14)$$

where

C_s = Concentration of contaminant in sediment (mg/kg)
S_s = Absorbed quantity of contaminant (portion to sediment) (mg)
X_s = Soil loss in runoff (kg)

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The contaminant concentration in the runoff effluent is:

$$C_e = M_s/V_r \tag{6-15}$$

where

C_e = Concentration of contaminant in runoff (mg/m^3)
M_s = Dissolved quantity of contaminant (mg)
V_r = Volume of runoff (m^3)

and

$$V_r = (CF)(SA)(Q_r) \tag{6-16}$$

where

CF = Conversion factor (100 for metric units)
SA = Contaminated surface area (hectares, ha)
Q_r = Depth of runoff (cm)

and

$$Q_r = (R_t - 0.2S_w)^2/(R_t + 0.8S_w) \tag{6-17}$$

where

R_t = Annual rainfall (cm)
S_w = Water retention factor (cm)

The contaminant concentration in the receiving water body downstream is:

$$C_w = (C_e)(q_p)/Q_t \tag{6-18}$$

where

C_w = Concentration of contaminant in water downstream (mg/m^3)
C_e = Concentration of contaminant in runoff (mg/m^3)
q_p = Peak runoff flow rate (m^3/sec)
Q_t = Flow rate of receiving water body (m^3/sec)

The numerical parameter values used to model the transport of soil by surface water runoff are application-specific. Modeling performed to date for operable unit risk assessments has utilized ranges of numerical values for model parameters for modeling contaminant concentrations in the surface water and sediment of the receiving water body. Parameter values for the USLE and

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MUSLE transport models will be determined on an operable unit-specific basis, and documented in the appropriate risk assessment document.

6.3 AIR TRANSPORT MODELING

Figure 6-5 depicts the modeling approach that will be used at the FEMP to estimate contaminant concentrations in air. Exposure concentrations of contaminants in air may be modeled for gaseous contaminants or particulate contaminants.

6.3.1 Particulate Contaminants

Estimating airborne concentrations of contaminants in the particulate phase involves modeling resuspension and dispersion. Resuspension of hazardous chemical and radionuclide contaminants may be estimated using a simple dust loading equation (DOE 1989) or resuspension rate model (Healy 1980) and the concentration of contaminants in surface soil available for resuspension (Figure 6-5). Dispersion may then be estimated using an air dispersion model such as AIRDOS-EPA (EPA 1979) to produce air concentrations at a variety of off-site locations, or a simple box model (GRI 1988) may be used to calculate air concentrations on site in the vicinity of the release point (Figure 6-5).

Alternatively, resuspension and transport of radioactive contaminants may be estimated for dose assessment purposes using the RESRAD model (DOE 1989) to calculate exposure concentrations of contaminants in air. The RESRAD model is also capable of modeling other exposure pathways for radioactive contaminants in soil. These uses are addressed in Sections 6.4 and 6.5.

6.3.1.1 Dust Loading Equation and Resuspension Rate Model

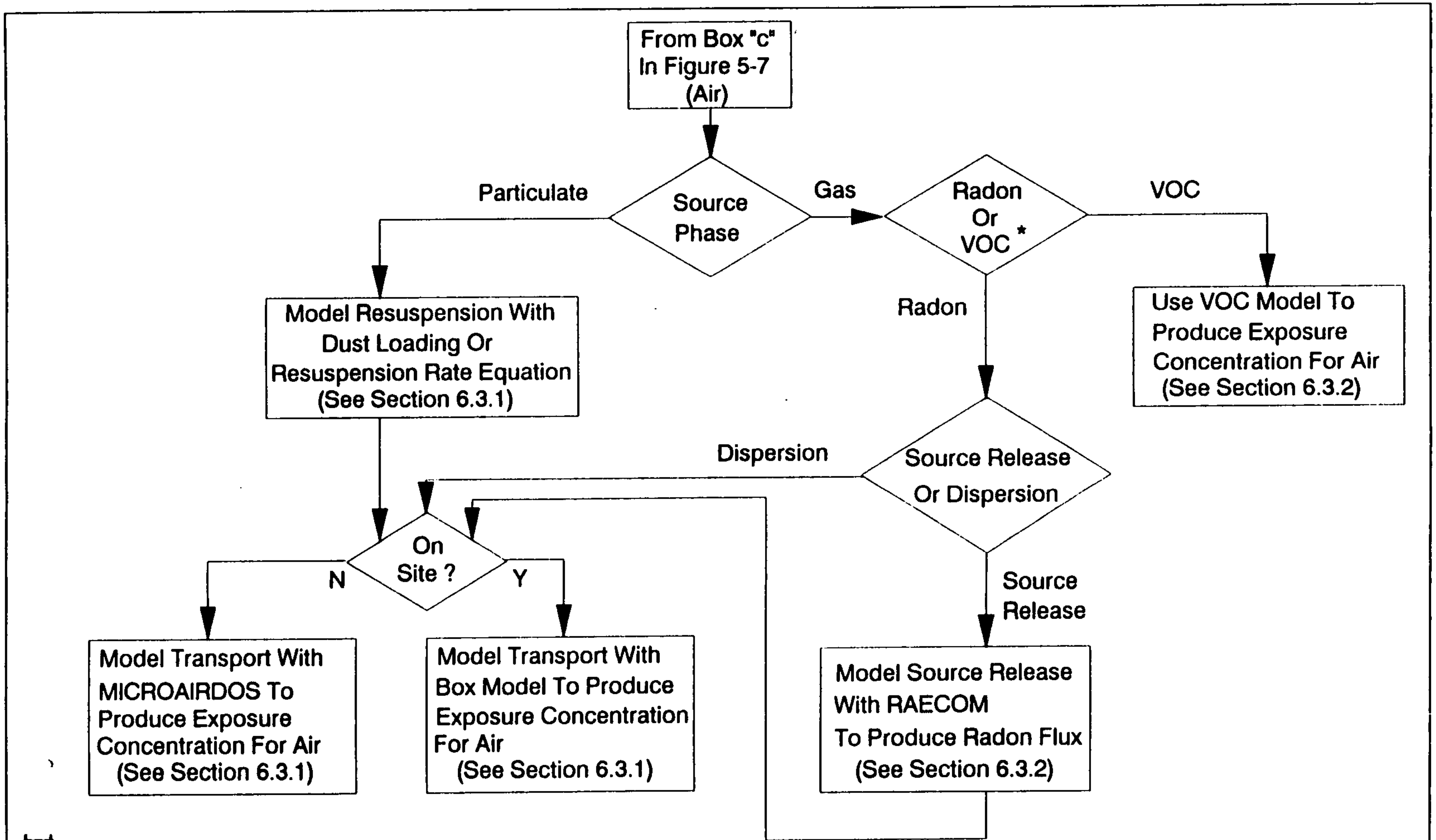
These methods are useful for estimating exposure concentrations of contaminants in air for workers involved in remediation activities at the contaminant release point. The dust loading equation used to estimate contaminant concentration in resuspended dust is based on the contaminant concentration in surface soil and a dust loading factor. The relationship is presented as (DOE 1989):

$$\text{(radionuclides)} \quad C_a = (D_l)(C_s) \tag{6-19}$$

$$\text{(chemicals)} \quad C_a = (D_l)(C_s)(CF) \tag{6-20}$$

where

C_a = Contaminant concentration in air (pCi/m^3); (mg/m^3)
D_l = Dust load factor (g of soil/m^3 of air)
C_s = Contaminant concentration in soil (pCi/g soil); (μg/g soil)
CF = Conversion factor (10^{-3} mg/μg)



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* VOC = Volatile Organic Compound

Figure 6-5 MODELING EXPOSURE CONCENTRATION FOR AIR

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Agricultural and remedial activities in the vicinity of the FEMP are expected to produce mechanical suspension of soil particles in air. The following dust loading factors (D_i) will be used as default values when site-specific data are not available:

Construction work	600 $\mu g/m^3$ [a]
Construction traffic	400 $\mu g/m^3$ [a]
Farming	200 $\mu g/m^3$ [b]
Other activities	100 $\mu g/m^3$ [c]

[a] DOE 1983
[b] DOE 1989
[c] NCRP 1984a

6.3.1.2 AIRDOS-EPA Model

Airborne transport of contaminated surface soils and gases is a pathway of concern at the FEMP. Therefore, it will be necessary to use a computer codes to calculate predicted concentrations of suspended and deposited contaminants at potential receptor locations.

The AIRDOS family of codes was selected to calculate expected concentrations of radiological constituents off site because site-specific data is available for them, and because past performance of these codes on the site is well documented. This family of codes includes AIRDOS-EPA (EPA 1979), which is typically run on a mainframe computer; and AIRDOS-PC (EPA 1989e) and MICROAIRDOS (Moore et al. 1989) which are suitable for use on personal computers. The first two, AIRDOS-EPA and AIRDOS-PC were selected because they have been approved for use in demonstrating compliance with 40 CFR 61.14. MICROAIRDOS has been conditionally approved to demonstrate compliance with NESHAPS Subpart H; National Emission Standards for Emissions of Radionuclides other than Radon from Department of Energy Facilities.

The AIRDOS-EPA family of codes uses a modified Gaussian plume to estimate horizontal and vertical dispersion of radionuclides released to the air. AIRDOS-PC reports radiation doses to humans while AIRDOS-EPA and MICROAIRDOS are capable of reporting:

- Concentrations in air
- X/Q values at receptor locations
- Rates of deposition on ground surfaces
- Ground surface concentrations
- Intake rates by man via food ingestion and air inhalation
- Radiation doses received by man

The parameter, X/Q, or "chi over que" is the calculated concentration of a contaminant in air at the location of interest per unit release of contaminant from a source as determined by atmospheric dispersion modeling. Values for X/Q are dependent on a number of factors, including release height, distance from source to receptor, wind speed and direction, and other meteorological conditions. The X/Q values reported by AIRDOS-EPA and MICROAIRDOS are necessary to calculate airborne concentrations of hazardous constituents at off-site receptor locations using the resuspension rate model equation (Healy 1980).

The model is defined as:

$$C_a = (R)(A)(X/Q) \qquad (6\text{-}21)$$

where

C_a = Air concentration downwind due to resuspension (pCi/m^3); (mg/m^3)
R = Resuspension rate (s^{-1})
A = Total quantity of contaminant in contaminated area (pCi); (mg)
X/Q = Atmospheric dispersion factor at the point of interest (s/m^3)

The total mass (A) of the contaminant in the contaminated area is defined as:

$$A = (C_p)(SA)(D_p)(\rho) \qquad (6\text{-}22)$$

where

A = Total quantity of contaminant in contaminated area (pCi); (mg)
C_p = Mean concentration of chemical in the contaminated area (pCi/kg); (mg/kg)
SA = Surface area available for wind resuspension (cm^2)
D_p = Depth of waste/soil available for wind resuspension (cm)
ρ = Density of waste/soil (kg/cm^3)

The resuspension rate, atmospheric dispersion factor and other parameters listed above are estimated on an operable unit-specific basis.

6.3.1.3 Box Model

A Nearfield Box Model (GRI 1988) may be used to calculate air concentrations on site adjacent to the release point. This method is useful for estimating exposure concentrations of contaminants in air for workers involved in remediation activities in the vicinity of contaminant release points. A box model requires little input information. For example, the contaminant release rate per unit surface area at the release point and the wind speed may be used, in conjunction with the mixing height, to estimate contaminant concentration in air in the vicinity of the release, as represented by:

$$C_a = (Q)/(H_b)(W_b)(U_m) \qquad (6\text{-}23)$$

where

C_a	=	Concentration of contaminant in ambient air on site (pCi/m^3) (mg/m^3)
Q	=	Emission rate of contaminant (pCi/sec) (mg/sec)
H_b	=	Downwind exposure height (m)
W_b	=	Width of crosswind dimension of contaminated area (m)
U_m	=	Average wind speed = $0.22\ (U_{10})\ \ln\ (2.5\ H_b)$ (m/sec)
U_{10}	=	Windspeed at 10 m above ground surface (m/sec)

and

$$Q = (J)(SA_c) \qquad (6\text{-}24)$$

where

J	=	Fluence rate ($pCi/m^2 \cdot sec$) ($mg/m^2 \cdot sec$)
SA_c	=	Contaminated area (m^2)

6.3.1.4 RESRAD Model

Resuspension and subsequent transport of radionuclide contaminants may be estimated using the most recent version of the RESRAD model (DOE 1989). The RESRAD model is capable of estimating potential exposures from all significant exposure pathways from contaminated soil or buried waste material. These exposure pathways include internal exposure from inhalation of airborne radionuclides in resuspended soil. RESRAD requires input of contaminant concentrations in surface material available for resuspension. A more complete discussion of the overall capabilities of RESRAD is presented in Section 6.6.

6.3.2 Gaseous Contaminants

Estimating airborne concentrations of contaminants in the gaseous phase such as volatile organic compounds (VOCs) and radon, involves modeling diffusion through media and dispersion in air following release. Airborne concentrations of VOC contaminants may be estimated using a simple VOC model to produce exposure concentrations in air (Figure 6-5). The transport model RAECOM (NRC 1984) will be used to model the release of radon from the surface of a radon source to the atmosphere, and the AIRDOS family of models (Section 6.3.1.2) or the box model (Section 6.3.1.3) will be used to model the subsequent transport of radon to off-site or on-site locations, respectively. The RAECOM model estimates the radon flux exiting the surface of source areas and cover material layers.

6.3.2.1 Volatilization Models

Volatilization and dispersion models used to estimate exposure to workers and to the public during remediation are presented below. These models are used to evaluate short-term effectiveness of remedial alternatives in the feasibility study, when VOCs are present in soil and soil excavation is a step in remedial alternatives. A VOC flux from soil is calculated using Equation 6-25, then air dispersion is modeled for on-site workers using the Nearfield Box Model, Equation 6-23. Final exposure concentrations to off-site residents are estimated using Equation 6-26.

Description of Models

Emission Rate Model (for waste at the saturated soil surface) (GRI 1988):

$$Q/SA_c = K_a\ (P - P_{inf})/(R)(T_p) \qquad (6\text{-}25)$$

where

Q/SA_c	=	Mass flux per unit area (moles/m^2 • hr)
SA_c	=	Contaminated surface area (m^2)
K_a	=	$0.0292\ (U^{0.78})(D_p^{-0.11})(Sc^{-0.67})$
U	=	Windspeed (m/hr)
D_p	=	Diameter of waste boundary (m)
Sc	=	Schmidt gas number (unitless)
P	=	Vapor pressure of the volatile at the soil surface (atm)
P_{inf}	=	Vapor pressure of the volatile in the atmosphere (atm)
R	=	Gas constant (atm • m^2/mol • °K)
T_p	=	Temperature of waste surface (°K)

The equation was modified to account for a mixture of volatiles present at less than saturation amounts by the factor C_i/C_s, where:

C_i	=	Measured concentration of a given volatile in soil (mg/kg)
C_s	=	Concentration if soil were saturated with a given volatile (mg/kg)

Dispersion of volatiles off site (Sector averaged model, zero stack height, GRI 1988):

$$C_a = (2\pi)^{1/2}[(8)(F_i)(Q)/(\pi)(\sigma_2)(U_m)(X)] \qquad (6\text{-}26)$$

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where

C_a	=	Concentration of contaminant in ambient air off site (mg/m^3)
F_t	=	Fraction of time wind is toward a given sector (unitless)
Q	=	Emission rate of contaminant (mg/sec)
σ_2	=	Standard deviation of crosswind concentration distribution (m)
σ_2	=	$(0.08)(1 + 0.0002X)^{-1/2}$
X	=	Distance from source (m)

These models make the following assumptions:

- Soils contaminated with VOCs will be excavated as part of the remedial alternative.
- An area of contaminated soils 10 m in diameter will be exposed at one time.
- VOCs will be present in a mixture of compounds. The average soil concentration for each area was used for C_i.

Parameter values for modeling the volatilization of organic compounds are presented in Table 6-5.

6.3.2.2 <u>RAECOM Model</u>

The migration of radon gas (radon-222) is modeled using the computer model RAECOM (NRC 1984). RAECOM is a radon generation and transport code that was originally designed to analyze radon generation and emanation through uranium mill tailings waste and earthen cover materials.

RAECOM is used in RI and FS risk assessments to analyze radon generation and emanation through media including waste materials at the FEMP, and cover materials such as soil, clay, and concrete. Media-specific parameter values are used. It is acknowledged that the use of a model for scenarios that are different from those for which it was originally designed introduces uncertainty in the results. Thus, the results will be used in operable unit RI and FS risk assessments with an appropriate level of caution.

RAECOM requires input of the thickness of each source material and cover material layer, the source strength expressed either as radium-226 concentration in the waste material or as radon flux exiting the surface of the waste material layer, and the porosity, moisture content, and radon gas diffusion coefficient for each source and cover material layer. The radon flux results are useful for comparison to radon flux criteria or for use in an air dispersion model.

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TABLE 6-5
VOLATILIZATION MODEL PARAMETER VALUES

Parameter	Value	Units	Reference
Sc - Schmidt number	Chemical-specific	(unitless)	TBD[a]
P - Vapor pressure	Chemical-specific	atm	TBD
P_{inf} - Partial pressure Infinite distance	0	atm	assumed
d - Liquid density	Chemical-specific	g/cm^3	TBD
C_i - Measured concentration	Chemical and Location -specific	mg/kg	from analytical results
U_m - Mean wind speed	16,600	m/hr	Dayton, OH; GRI, 1988
D_p - Diameter of site boundary	Location-specific	m	TBD
A_c - Surface area	Area dependent	m^2	calculated from Dp
T_p - Surface temperature	293	°K	20° C
E - Soil porosity	0.3	(unitless)	average for fine sand; GRI 1988
D - Soil density	1.7	g/cm^3	average for FEMP
H_b - Downwind height of box	1.83	m	assuming a worker height of 6 feet
W_b - Width of box	Location-specific	m	TBD
R - Universal gas constant constant	8.21×10^{-5}	$atm \bullet m^3/mol \bullet °K$	universal gas
F_t - Frequency of wind direction	Location-specific	(unitless)	estimated from local wind data
X - Distance from source	Location-specific	m	TBD

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[a] TBD - To be determined, based on specific applications.

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RAECOM calculates the radon flux exiting the surface of the upper layer of cover material. The code is based on a one-dimensional, multilayer solution of Fick's law using the boundary conditions set forth in NUREG/CR-3533 (NRC 1984). For a bare source, this solution becomes:

$$J_t = (10^4)(R)(\rho_t)(E)[(\lambda)(DC_t)]^{1/2} (\tanh [(x_t)(\lambda/DC_t)^{1/2}]) \qquad (6\text{-}27)$$

and for a covered source the solution is:

$$J_c = \frac{(2)\,(J_t)\,(e^{-b_c x_c})}{[1 + ((a_t/a_c)^{1/2})\,(\tanh(b_t x_t))] + [1 - ((a_t/a_c)^{1/2}\,(\tanh(b_t x_t))]\; e^{-2 b_c x_c}} \qquad (6\text{-}28$$

where

J_t	= Radon flux from the source materials surface (pCi/m^2-sec)	
R	= Specific activity of radium in the source materials (pCi/g)	
ρ_t	= Dry bulk density of source material (g/cm^3)	
E	= Radon emanation coefficient (unitless)	
DC_t	= Radon diffusion coefficient in the total tailings pore space (cm^2/sec)	
λ	= Radiological decay constant of radon (2.1 x 10^{-6} sec^{-1})	
J_c	= Radon flux from the surface of cover material (pCi/m^2/sec)	
b_c	= $(\lambda/DC_c)^{1/2}$ (cm^{-1})	
x_c	= Thickness of cover material (cm)	
a_t	= $(\rho_t)^2(DC_t)\,[1 - (1\text{-}k)\,m_t]^2$ (cm$_2$/sec)	(6-29)
a_c	= $(\rho_c)^2(DC_c)\,[1 - (1\text{-}k)\,m_c]^2$ (cm$_2$/sec)	(6-30)
b_t	= $(\lambda/DC_t)^{1/2}$ (cm^{-1})	
x_t	= Thickness of tailings (cm)	
DC_c	= Radon diffusion coefficient in the total cover pore space (cm^2/sec)	
m	= Fractional moisture saturation (unitless)	
k	= Radon distribution coefficient, C/C (unitless)	
ρ_c	= Dry bulk density of cover (g/cm^3)	

Care must be taken when applying this code to multilayer systems. Due to the boundary conditions selected, the code may be unable to analyze the radon flux passing from a high density material to a material with a much lower density in some systems with more than two layers. (See Equations A-6 and A-7 in Appendix A of NRC 1984).

The RAECOM code requires a limited amount of information to estimate radon flux (pCi/m^2-sec) from the surface of a radon source layer and cover materials. Necessary information includes either the radium-226 concentration in source material or radon flux from the source material; plus the thickness, porosity, moisture content, and diffusion coefficient for each layer of source or

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cover material included in the model application. Values for these parameters vary among operable units. The parameters and the range of values used to assess radon emanation are listed below:

Parameter	Value Range	Units	References
Soil (Cover)			
Porosity	0.30	unitless	Assumption
Moisture	13 - 40	% dry wt	IT 1991
Radium Concentration	1.5	pCi/g	Myrick 1983
Diffusion Coefficient	0.03 - 0.04	cm^2sec^{-1}	RAE 1990, NRC 1984
Concrete (Cover)			
Porosity	0.05 - 0.25	unitless	Culot 76, Assump.
Moisture	0 - 15.7	% dry wt	Assump., calc'd
Radium Concentration	0	pCi/g	Assumption
Diffusion Coefficient	1.69E-5 - 3.0E-3	cm^2sec^{-1}	RAE 1990, NRC 1984, Culot 1976
Untreated Waste (Source)			
Porosity	0.30	unitless	Assumption
Moisture	13 - 40	% dry wt	IT 1991
Radium Concentration	operable unit-specific	pCi/g	RAE 1990, NRC
Diffusion Coefficient	0.04	cm^2sec^{-1}	1984, Culot 1976
Treated Waste (Source)			
Porosity	0.25 - 0.3	unitless	Culot 1976, Assump.
Moisture	0 - 15.7	% dry wt	Assump, calc'd
Radium Concentration	operable-unit specific	pCi/g	RAE 1990, NRC
Diffusion Coefficient	1.69E-5 - 3.0E-3	cm^2sec^{-1}	1984, Culot 1976

6.4 FATE OF CONTAMINANTS IN SOIL

Figure 6-6 depicts the technical approach that will be used to estimate contaminant concentrations in soil. Modeling exposure concentrations of contaminants in soil for soil exposure pathways requires characterization of the soil source term. This characterization must include identification of contaminants in the soil, estimation of the quantity or concentration of 152 contaminants in the soil, determination of the quantity of soil potentially available to interact in exposure pathways, and estimation of soil properties that are pertinent to modeling contaminant transport and receptor exposure to contaminants.

Given adequate characterization or estimation of contaminant concentrations in soil that may potentially be involved in receptor exposures, the soil ingestion exposure pathway leads directly to the intake assessment process (Figure 6-6) without any modeling of contaminant transport. Other direct exposure pathways include dermal contact with skin (see Section 7.2.1.7) and direct exposure to penetrating radiation (Section 6.5).

Remaining exposure pathways in Figure 6-6 require modeling the contaminant transport from soil to other environmental media. These types of transport modeling required includes modeling the leaching of contaminants from soil to the aquifer (Section 6.1), modeling the erosion of contaminants from soil to surface water bodies and stream beds (Section 6.2), and modeling the resuspension of contaminants from soil to the air (Section 6.3).

6.5 MODELING DIRECT RADIATION EXPOSURE

Direct radiation exposure can be quantitatively evaluated via modeling when direct radiation exposure measurement data are not available. A number of risk assessment scenarios in operable unit baseline and FS risk assessments require that penetrating gamma radiation dose rates from waste sources be calculated. In addition, modeling is used to estimate baseline dose rates from portions of the FEMP that lack characterization for penetrating gamma radiation. For example, modeling is used to estimate dose rates from waste shipments proposed as part of remedial alternatives that involve transportation of waste to a disposal facility. Modeling is also used to estimate penetrating gamma radiation dose rates to remediation workers during phases of cleanup that involve excavation or removal of waste material that is a source of significant gamma radiation fields.

In order to apply a model to estimate direct radiation exposure, the source geometry must be identified, including consideration of the presence of shielding between the radiation source and the receptor (Figure 6-7). The figure illustrates selection of planar source geometry or a nonplanar source geometry.

Radiation dose rates for planar source geometries that do not involve shielding materials may be modeled using either RESRAD (DOE 1989) or MICROSHIELD (Grove 1988) (Figure 6-7). The most common example of this scenario at the FEMP is irradiation by radionuclides in planar areas of contaminated surface soil. This exposure pathway applies to receptors such as the

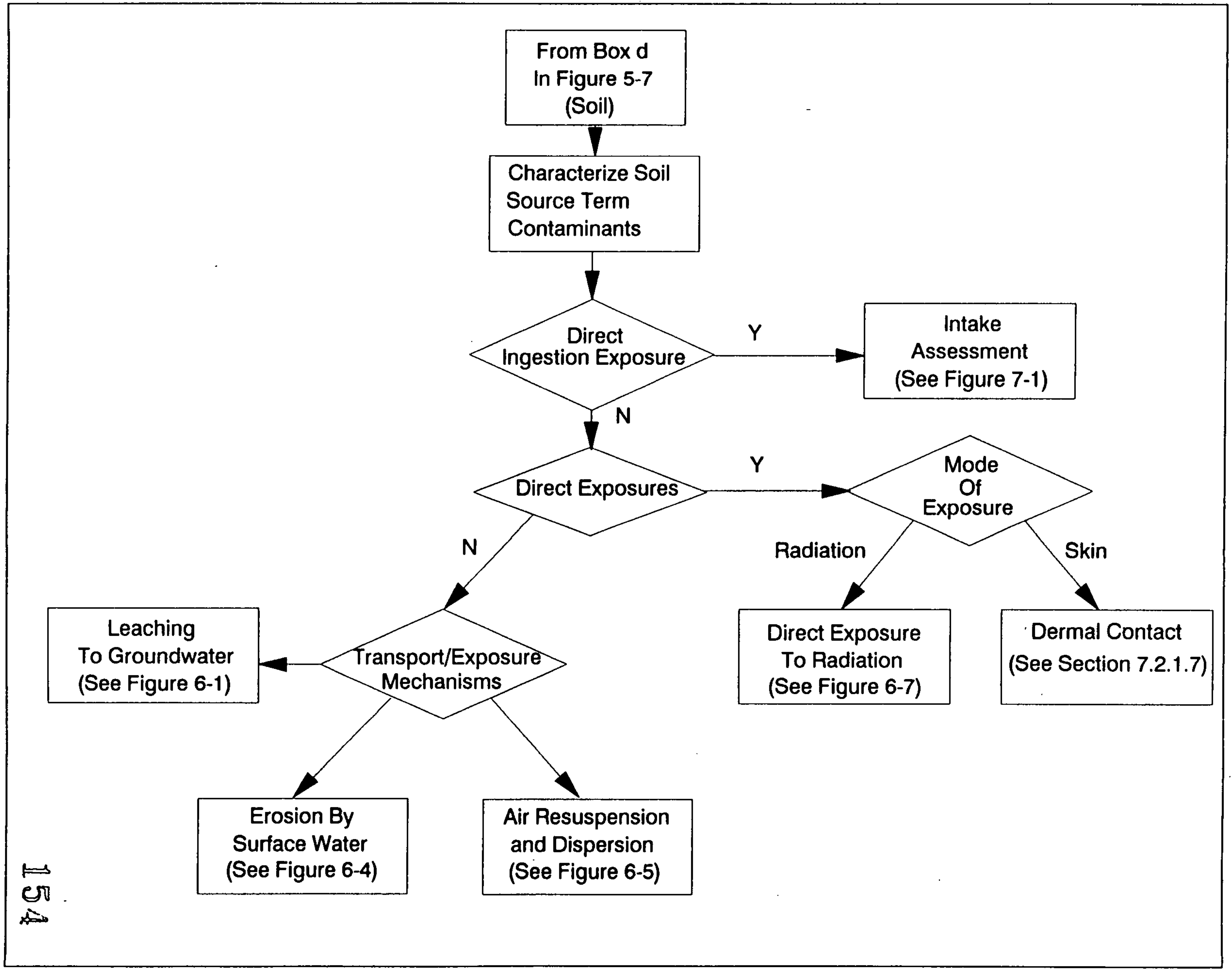

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Figure 6-6 MODELING EXPOSURE CONCENTRATION FOR SOIL

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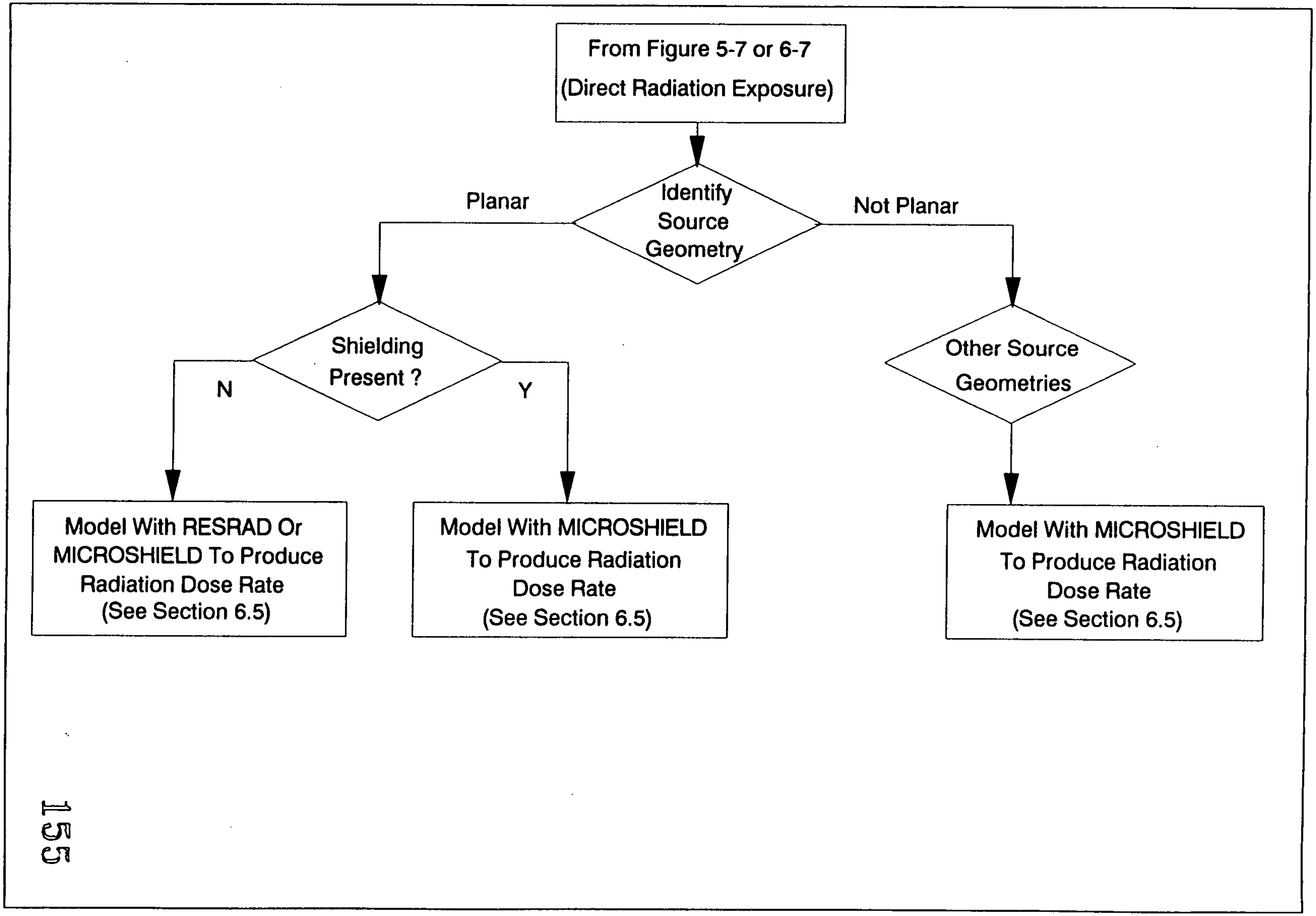

Figure 6-7 MODELING DOSE RATE FOR DIRECT RADIATION EXPOSURE

resident farmer, some of the remediation workers, intruders in contaminated areas, and individuals that may be exposed during transportation of radiation-emitting waste materials to a disposal site. As stated in Section 6.6, the RESRAD code is capable of estimating potential exposures from direct radiation exposure from penetrating radiation.

Radiation dose rates for planar source geometries that involve shielding materials and for nonplanar source geometries are modeled using the MICROSHIELD 3.0 code (Grove 1988). MICROSHIELD was developed for use on personal computers by Grove Engineering (Grove 1988), and uses the same algorithms as ISOSHLD, a mainframe code developed by Battelle Northwest Laboratories (Engle 1966). MICROSHIELD offers a variety of source geometries that are used in RI/FS risk assessments to suit operable unit specific modeling needs.

MICROSHIELD methodology offers a tested approach for estimating the dose rate to an individual from external gamma radiation. MICROSHIELD presents the estimated dose rate from a given configuration in three forms; activity (photons/sec), gamma flux energy density (MeV/cm^2-sec), and dose rate (mrad/hr). The program requires a moderate amount of information to perform these analyses. Most input parameters are simple to determine, but care must be taken when determining the most appropriate source geometry and shielding configurations. Basic information requirements can be grouped into three categories: source term configuration, shielding arrangement, and receptor/detector placement. These three information groupings are described below.

MICROSHIELD uses information on the gamma source composition, geometry, and orientation to calculate the energies and fluxes of the gamma radiation leaving the source. The composition of the source is characterized by information on the types and densities of the source materials, and the types and concentrations of nuclides in the source. The code uses this information, and data on the source geometry and orientation with respect to the location of the receptor, to calculate the gamma-ray flux density emitted in the general direction of the receptor. Information on any materials between the source and the receptor allows the code to calculate the degree to which the gamma rays emitted by the source are attenuated by the intervening material. In addition, the code can use information on the chemical and physical properties of the shielding and source materials to estimate any additional exposure caused by scattering phenomena (buildup).

Receptor placement determines the thickness of the air gap between the receptor and the last shield. This is potentially important because the air gap provides additional shielding and gamma exposures decrease as a function of distance from the source.

The source/shielding configurations used to represent the exposure scenarios modeled vary considerably between operable units. Other geometries may be identified for external radiation exposure assessments of FEMP risk assessments. Parameter values selected for subsequent risk assessment modeling needs may vary.

6.6 MULTIPLE PATHWAY ASSESSMENT CODES

A multi-pathway code calculates the combined doses to a receptor from multiple pathways at the same time. These codes have the advantage of being able to account for simultaneous time-dependent source depletion by more than one pathway. For example, contaminants leached to the groundwater will be subtracted from the total source available for surface erosion in the next time increment.

RESRAD (DOE 1989a) is an example of a multi-pathway computer code that is used to perform exposure assessments for complex sites that potentially involve numerous interacting pathways. Other comparable computer codes exist, which can be used in place of or in conjunction with RESRAD. Examples include PRESTO (EPA 1989d), PATHRAE (DOE 1986a, DOE 1986b) and GENII (DOE 1988c, DOE 1988d, DOE 1990). Unfortunately, none of these codes incorporate EPA's HEAST methodology at this time, so their use in FEMP RI/FS risk assessments is restricted to dose assessment.

Because the pathways evaluated in RESRAD are not identical to those presented by this addendum, RESRAD is only suitable for limited dose assessment applications at the FEMP. The computer code is capable of estimating potential exposures from all significant exposure pathways from contaminated soil. These pathways include:

- Direct exposure to penetrating radiation from contaminated soil
- Internal exposure from inhalation of airborne radionuclides in resuspended soil
- Internal exposure from ingestion of:
  - Plant foods grown in contaminated soil
  - Meat and milk from livestock fed with contaminated feed and water
  - Drinking water from a contaminated well
  - Fish from a contaminated pond

RESRAD uses a pathway analysis method involving predicted relationships (media transfer factors) between radionuclide concentrations in the different media which make up each of the pathways listed above. Ultimately, these media transfer factors are combined into one factor (the concentration factor) relating the radionuclide concentration in soil to radiation dose.

Concentrations of a given radionuclide in food crops and livestock feed are derived by multiplying the nuclide's elemental soil-to-plant transfer factor by its calculated or measured concentration in soil.

Concentrations of radionuclides in meat and milk are derived by multiplying their elemental plant-to-meat or plant-to-milk transfer factors by the cow consumption rate of feed. Equations for the pathway concentration factors and media transfer factors associated with each pathway are presented and discussed in detail in the manual accompanying the RESRAD code (DOE 1989). This extensive and detailed material is not reproduced in this work plan addendum.

The numerical values for human intake and agricultural parameters input into RESRAD will be made consistent with those selected for corresponding transport, intake and exposure model equations presented in Section 6.0 and Section 7.0. Where possible, model equations will comply with the equations in this work plan. Variances in equations used will be documented and presented to EPA along with the projected impacts of those variances.

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7.0 QUANTIFICATION OF EXPOSURE

This section contains a description of the methodology used to quantify both long- and short-term exposures for exposure pathways of concern at the FEMP. This methodology employs the concept of the Reasonable Maximum Exposure, or "RME." The RME is the maximum exposure reasonably expected to occur at the site (EPA 1989a). If the RME is determined to be acceptable, then it is likely that all other lesser exposures at the site will also be acceptable.

The methodology discussed includes the approach for determining exposure concentration(s) at a given location (Section 7.1), the exposure models used to quantify any resulting intakes (Section 7.2), and the methodology to be used to quantify ecological effects of exposures to the contaminants present at the FEMP (Section 7.3).

7.1 DETERMINATION OF EXPOSURE CONCENTRATION

The exposure concentration is the concentration of a contaminant in an exposure medium that will be contacted by a real or hypothetical receptor. Determination of the exposure concentration depends on factors such as:

- Availability of data from which an exposure concentration can be determined
- Statistical methodologies selected to determine the appropriate exposure concentration
- Potential contributions to contaminant concentration from background concentrations not attributed to the site
- Potential contribution to contaminant concentrations from contaminants attributable to other operable units
- Location of the potential receptor

Exposure concentrations at the FEMP will be determined in two different ways. When sufficient analytical data are available, measured concentrations are used. When the quality or quantity of data is insufficient, consideration is given for obtaining better or additional data. If additional measurement data cannot be obtained, modeled concentration data will be used. This section addresses the methodologies used to derive exposure concentrations from the two types of data.

7.1.1 Measured Concentrations

When analytical results are available, these data will be used to determine the appropriate receptor exposure concentration for current exposure pathways. Data from the sources listed in Section 3.0 will be used to assemble these data sets.

To be consistent with the concept of the RME scenario, an estimate of the highest exposure that can reasonably be expected to occur at the FEMP will require a reasonable maximum estimate of the concentration of each contaminant in each exposure medium. Because of the uncertainty associated with any estimate of exposure concentrations, the upper 95% confidence limit on the arithmetic mean for either a normal or lognormal distribution is the recommended statistic (concentration value) to be constructed from measured contaminant concentration data and used in subsequent risk assessments (EPA 1991e). This term is generally called the upper confidence limit (UCL) and will be used as the representative exposure concentration derived from measured data at the FEMP.

In order to construct the UCL, a determination of the distribution type (normal, lognormal, or other) must be made. The methodology for determining the distribution type for site-related data is the same as the methodology for background data described in Section 4.2.1. The minimum number of site-related data values necessary to adequately determine the distribution type is arbitrarily chosen as twelve (12), of which at least 50% exceed the SQL. Data reported as non-detects will be assigned a value of ½SQL for the purpose of calculating the UCL.

Data sets having fewer than the minimum number of measurements for determining the distribution will be evaluated on a case-by-case basis. Generally, the highest measured concentration will be used as the exposure point concentration for a data set (EPA 1991e) for which the distribution type cannot be determined.

Site-related data sets will be evaluated for the presence of outliers with the methods described in Section 4.2.3. The potential causes of outliers will be investigated. When outliers cannot be attributed to errors, they will be included in the calculation of exposure point concentrations.

The UCL will be calculated for a normal distribution as follows:

$$UCL = \bar{x} + t_{1-\alpha,\, n-1} \bullet (s/\sqrt{n}) \qquad (7\text{-}1)$$

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TABLE 7-1
CRITICAL VALUES FOR STUDENT'S t-DISTRIBUTION[a]

n-1	$t_{0.95, n-1}$	n-1	$t_{0.95, n-1}$
1	6.314	16	1.746
2	2.920	17	1.740
3	2.353	18	1.734
4	2.132	19	1.729
5	2.015	20	1.725
6	1.943	21	1.721
7	1.895	22	1.717
8	1.860	23	1.714
9	1.833	24	1.711
10	1.812	25	1.708
11	1.796	30	1.697
12	1.782	40	1.684
13	1.771	60	1.671
14	1.761	120	1.658
15	1.753	∞	1.645

[a] (Koopmans 1987)

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where

$\bar{x}$	=	sample arithmetic mean
$t_{1-\alpha, n-1}$	=	critical value for Student's t-Distribution (given in Table 7-1)
α	=	0.05 (i.e., $1-\alpha = 0.95$ or 95% confidence limit for a one-tailed test)
n	=	number of samples in the set
s	=	sample standard deviation

The UCL will be calculated for a lognormal distribution as follows:

$$UCL = e^{\bar{y} + \frac{1}{2} s_y^2 + H_{0.95} \cdot s_y / (n-1)^{\frac{1}{2}}} \qquad (7\text{-}2)$$

where

$\bar{y}$	=	$\sum y/n$ = sample arithmetic mean of the log-transformed data, $y = \ln x$
s_y	=	sample standard deviation of the log-transformed data
n	=	number of samples in the data set
$H_{0.95}$	=	value for computing the one-sided upper 95% confidence limit on a lognormal mean from standard statistical tables (Gilbert 1987)

The 95% confidence limit on the arithmetic mean for the background concentration for each carcinogen (including radionuclides) will be subtracted from the site-related UCL for the carcinogen to determine exposure concentrations of carcinogens at exposure points. In this way the quantified exposure and risks that represent the excess attributable to contamination from the site can be presented. In addition, exposures to background concentrations of carcinogens (including radionuclides) will be assessed to provide the risks associated with exposures that are not attributed to the site. This information facilitates the important comparison of the background risks, the added risks due to the site, and the total risk (background risk plus risk from the site).

Background concentrations of chemical toxicants will not be subtracted from UCL values when determining exposure point concentrations. Thus, the quantified exposure and risk represent that which is attributable to contamination from the site plus background.

7.1.2 Modeled Concentrations

When analytical results are not available, a model must be used to predict potential exposure concentrations. For example, a quantitative assessment of future potential exposures will depend

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on predicted concentrations. It may also be necessary to model exposure concentrations at potential receptor locations for current exposure pathways if measured analytical data are unavailable or insufficient for quantifying the RME. Model source terms are constructed using the 95% confidence limit on the arithmetic mean of the site-related concentrations.

The UCL will not be constructed for concentrations determined by modeling for the FEMP. The RME scenario for modeled data will assume that the hypothetical receptor is at the location having the reasonable maximum contaminant concentration. This location will be determined by quanititative means for groundwater modeling, and by concentration/toxicity/access screening for models for other media. For multiple contaminants and pathways, the hypothetical receptor will be assumed to be at the location having the reasonable maximum total risk from all contaminants and pathways. These concentrations will be calculated using the models and methodologies detailed in Section 6.0. The above-background concentrations will then be used in the remainder of the exposure assessment and risk assessment.

7.2 INTAKE ASSESSMENT

The methodologies and parameter values that will be used to quantitatively estimate contaminant intakes for the RI and FS human health risk assessments at the FEMP are presented in this section. In general, the magnitude of contaminant intake depends on the route of exposure and the variables impacting the transmittal of contaminants via that route. These intake estimates will be used in conjunction with contaminant toxicity data to quantify the risks associated with the RME for each pathway.

Quantitative intake assessments will be performed for all plausible intakes of contaminants by humans in the RI and FS exposure assessments. The models and equations presented in this section have been obtained from EPA risk assessment guidance (EPA 1989a). Additional models presented in the U.S. Nuclear Regulatory Commission (NRC) Regulatory Guide 1.109 (NRC 1977) will be used for situations not specifically addressed in the EPA risk assessment guidance. Examples of such situations are given in this section.

The RI/FS at the FEMP is being managed as five operable units with separate baseline risk assessments, a Preliminary Site-Wide Baseline Risk Assessment, a Site-Wide RI/Projected Residual Risk Assessment, FS risk assessments for each operable unit, and a Site-Wide FS/Risk Assessment. Maintaining a high level of consistency among operable unit risk assessments and site-wide risk assessments is desired. For example, it is generally appropriate to quantify contaminant exposures of a similar receptor, through the same pathway, in the same manner for each operable unit. However, at times unique scenarios and circumstances occur that lead to justifiable differences in the process of estimating exposure. For example, variation in the level of characterization available for different portions of the site may justify using different assumptions and parameter values (if available) for modeling exposures from different portions of the site. Justification for use of different assumptions and parameter values will be presented in each risk assessment. Therefore, the exposure assessments conducted for operable unit baseline risk assessments, site-wide risk assessments, and FS risk assessments may not be identical.

The exposure assessment models and most of the parameters presented in this section are used in one or more (but not necessarily all) of the baseline or FS risk assessments. The relationships among models are noted as appropriate.

The method for estimating the committed effective dose equivalent (CEDE) from intake of radionuclides is also included in this section. Estimated CEDEs are used because they will be compared to pertinent radiation dose limits. The method for estimating injuries and fatalities from construction and transportation accidents for FS risk assessments is also presented in this section.

The intake assessment process is illustrated in Figure 7-1. A quantitative estimate of contaminant intake is determined and the intake assessment process is applied to an exposure scenario. Figure 7-1 depicts receptor exposure mechanisms including inhalation, ingestion, and dermal contact. Each exposure mechanism in Figure 7-1 leads to the subsections of Section 7.2.1 and specifies the models used to quantify receptor intake.

7.2.1 Intake Models and Equations

Each intake model equation that corresponds to ingestion or inhalation by an adult generates a calculated intake of radioactive material (picocuries [pCi]) and a daily chemical intake per unit body weight (mg/kg-day). Model equations that do not correspond to an adult intake produce calculated contaminant concentrations in intermediate media such as vegetables, forage, meat, and milk. Spreadsheets are used for calculations of intake, cancer risks, and radiation doses. Parameter values used in FEMP RI/FS risk assessments for intake and exposure calculations are presented in Section 7.2.2.

Section 7.2.1.6 describes the fish ingestion model equation. Sections 7.2.1.7 and 7.2.1.9 address dermal contact and penetrating radiation exposure pathways.

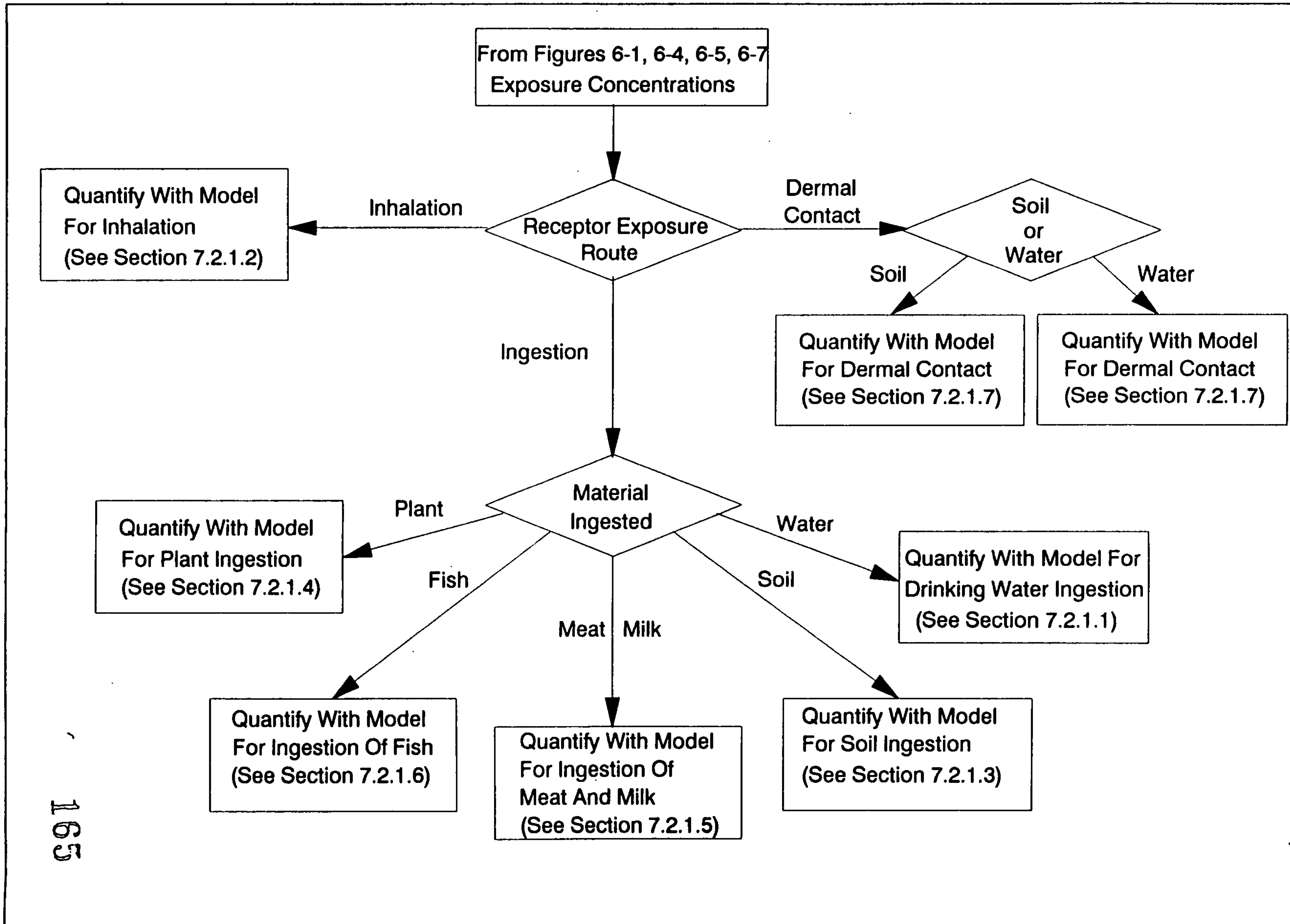

Figure 7-1 INTAKE ASSESSMENT

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7.2.1.1 Drinking Water

The equations used to estimate intake from drinking water are adapted from EPA (EPA 1989a). For variables that are common to both chemical and radionuclide intake equations, units for the radionuclide equations are listed first. The intake equations are:

$$\text{(radionuclides)} \quad I_w = (C_w)(IR)(ED) \qquad (7\text{-}3)$$
$$\text{(chemicals)} \quad I_w = (C_w)(IR)(ED)(EF)/(BW)(AT) \qquad (7\text{-}4)$$

where

I_w = Intake from drinking water (pCi) (mg/kg-day)
C_w = Concentration in water (pCi/L) (mg/L)
IR = Ingestion rate (L/yr) (L/day)
EF = Exposure frequency (day/yr)
ED = Exposure duration (yr)
BW = Body weight (kg)
AT = Averaging time (days); for noncarcinogens, AT equals (ED)(350 days/yr [EPA 1991c]); for carcinogens, AT equals (70-year lifetime)(365 days/yr)

7.2.1.2 Inhalation

The equations used to quantify intake from the inhalation pathway adapted from EPA (EPA 1989a) are:

$$\text{(radionuclides)} \quad I_a = (C_a)(IR)(ED) \qquad (7\text{-}5)$$
$$\text{(chemicals)} \quad I_a = (C_a)(IR)(EF)(ED)/(BW)(AT) \qquad (7\text{-}6)$$

where

I_a = Intake from inhalation (pCi) (mg/kg-day)
C_a = Concentration in air (pCi/m^3) (mg/m^3)
IR = Inhalation rate (m^3/yr) (m^3/day)
EF = Exposure frequency (day/yr)
ED = Exposure duration (yr)
BW = Body weight (kg)
AT = Averaging time (days); for noncarcinogens, AT equals (ED)(350 days/yr); for carcinogens, AT equals (70-year lifetime)(365 days/yr)

The estimation of intake of contaminants in soils through the inhalation of fugitive dust may be determined using the concentration of contaminants in soil at the RME location. The methods for quantifying contaminant concentrations in dust are presented in Section 6.3.

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7.2.1.3 <u>Ingestion of Soil/Sediment</u>

The estimation of intake of contaminants in soils or sediment is determined using the concentration in the soil or sediment at the RME location. Evaluation of the soil and sediment ingestion pathway is performed for adults and children. Children represent a critical subpopulation for whom these exposure pathways may be significant. EPA guidance suggests that children may be exposed through the soil ingestion pathway at ages 1 through 6 (EPA 1989a). It is assumed that ingestion of sediments in stream beds away from the home involves slightly older children at ages 6 through 17. The equations used to quantify intake (EPA 1989a) are:

$$\text{(radionuclides)} \quad I_s = (C_s)(IR)(ED)(EF)(FI) \qquad (7\text{-}7)$$
$$\text{(chemicals)} \quad I_s = (C_s)(IR)(CF)(FI)(EF)(ED)/(BW)(AT) \qquad (7\text{-}8)$$

where

I_s = Intake from soil or sediment (pCi) (mg/kg-day)
C_s = Concentration in soil or sediment (pCi/g) (mg/kg)
IR = Ingestion rate (g/day) (mg/day)
CF = Conversion factor 10^{-6} kg/mg
FI = Fraction ingested from contaminated source (unitless)
EF = Exposure frequency (days/yr)
ED = Exposure duration (yr)
BW = Body weight (kg)
AT = Averaging time (equals ED x 350 days/yr) (days)

7.2.1.4 <u>Ingestion of Vegetables</u>

Currently, irrigation of farm land in the vicinity of the FEMP is not widely practiced. In Hamilton and Butler counties, an average of less than 1.5 percent of farmland is irrigated (Bureau of Census 1989):

	Hamilton County	Butler County
Irrigated acres -	676	362
Total farm acres -	28,318	159,519
% land irrigated -	2.4%	0.2%

However, ingestion of farm and homegrown products irrigated with contaminated groundwater or surface water is evaluated in the FEMP risk assessments because of the potential for this to become a viable pathway at any time in the near future, and because reported statistics may not reflect potential irrigation of home gardens.

The equations used to estimate exposure to chemicals and radionuclides via ingestion of vegetables irrigated with contaminated water are from the NRC (NRC 1977) and the EPA (EPA

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1989a). The two-step process first involves the calculation of the concentration of the contaminant on and in the plant as a result of foliar deposition and root uptake, followed by the calculation of intake from consumption of the plant by humans. The model used to estimate the concentration in and on vegetation irrigated with contaminated water is (NRC 1977):

$$C_{ivw} = d_w \left[\frac{I_w (1 - e^{-\lambda_{Ei} t_e})}{Y \lambda_{Ei}} + \frac{f_w B_{iv(1)} (1 - e^{-\lambda_{ri} t_{bw}})}{\rho \lambda_{ri}} \right] e^{-\lambda_{ri} t_h} \qquad (7\text{-}9)$$

For vegetation exposed to atmospheric fallout of dust, the equation becomes (NRC 1977):

$$C_{ivd} = d_d \left[\frac{I_d (1 - e^{-\lambda_{Ei} t_e})}{Y \lambda_{Ei}} + \frac{f_d B_{iv(1)} (1 - e^{-\lambda_{ri} t_{bd}})}{\rho \lambda_{ri}} \right] e^{-\lambda_{ri} t_h} \qquad (7\text{-}10)$$

where

- λ_{Ei} = Effective depletion constant of i^{th} contaminant on the surface plants (hr^{-1})
- λ_{ri} = Radioactive or chemical decay constant of i^{th} contaminant (hr^{-1})
- $B_{iv(1)}$ = Dry soil to wet plant partitioning coefficient of i^{th} contaminant (C_{iv}/C_s)
- C_{ivd} = Concentration of i^{th} contaminant in plants as a result of deposition of contaminated dust on plants (pCi/kg) (mg/kg)
- C_{ivw} = Concentration of i^{th} contaminant in plants as a result of irrigating plants with contaminated water (pCi/kg) (mg/kg)
- d_d = Dust deposition rate (pCi/m^2-hr) (mg/m^2-hr)
- d_w = Irrigation deposition rate (pCi/m^2-hr) (mg/m^2-hr)
- f_d = Fraction of year plant is irrigated (unitless)
- f_w = Fraction of year plant is downwind (unitless)
- ρ = Effective dry surface density of the soil (kg/m^2)
- r_d = Fraction of deposited dust retained on plant surface (unitless)
- r_w = Fraction of water borne material retained on plant surface (unitless)
- t_{bd} = Duration of facility operation (hrs)
- t_{bw} = Duration of irrigation use (hrs)
- t_e = Growing season (hrs)
- t_h = Duration of period between harvest and consumption (hrs)
- Y = Agricultural yield (kg/m^2)

In addition to exposure to contaminated irrigation water and dust, vegetables and livestock feed may be contaminated by root uptake from contaminated soil or waste. A contribution via this pathway is accounted for in the irrigation model; however, this pathway is also considered for areas that are not irrigated with contaminated water but that exhibit surface soil contamination from historical deposition on the soil by various means. The following equation can be used to calculate the contaminant concentration in the plant from root uptake of contaminants already in the soil.

$$C_{ivs} = (C_s)\,(B_{iv(1)})\,(e^{-\lambda_{ri}(t_e+t_h)}) \qquad (7\text{-}11)$$

where

C_{ivs} = Concentration of i^{th} contaminant in plants as a result of root uptake from contaminated soil (pCi/kg) (mg/kg)
C_s = Concentration of i^{th} contaminant in dry soil at the beginning of the growing season (pCi/kg) (mg/kg)

The total concentration of contaminants in vegetables (C_{iv}) is calculated with the following equation:

$$C_{iv} = C_{ivw} + C_{ivd} + C_{ivs} \qquad (7\text{-}12)$$

Once the concentration in vegetation has been determined, intake can be calculated with the following equations:

$$\text{(radionuclides)} \quad I_{iv} = (C_{iv})(IR)(ED)(FI) \qquad (7\text{-}13)$$
$$\text{(chemicals)} \quad I_{iv} = (C_{iv})(IR)(FI)(EF)(ED)/(BW)(AT) \qquad (7\text{-}14)$$

where

I_{iv} = Intake from vegetation (pCi) (mg/kg-day)
C_{iv} = Total concentration of contaminants in vegetable (pCi/kg) (mg/kg)
IR = Ingestion rate (kg/yr) (kg/day)
FI = Fraction ingested from contaminated source (unitless)
EF = Exposure frequency (days/yr)
ED = Exposure duration (yr)
BW = Body weight (kg)
AT = Averaging time (days); for noncarcinogens, AT equals (ED)(350 days/yr); for carcinogens, AT equals (70-year lifetime)(365 days/yr)

Equations of the same form are used to determine the contaminant concentration in livestock feed, substituting concentration factors for livestock feed in place of those for vegetables consumed by man. Once the contaminant concentrations in vegetables and livestock feed have been determined, intake can be estimated using the intake equations presented for ingestion of vegetables contaminated by irrigation and ingestion of animal products.

7.2.1.5 Ingestion of Animal Products

As in the quantification of intake following exposure to vegetables, the concentration in animal products must be estimated prior to the determination of intake. The concentration of a contaminant in animal products, such as beef or milk, is determined using the following equation (NRC 1977):

$$C_{iA} = F_{iA}[(C_{if})(Q_f) + (C_{iAw})(Q_{Aw})] \qquad (7\text{-}15)$$

where

C_{iA} = Concentration of i^{th} contaminant in the animal product (pCi/L for milk, pCi/kg for beef) (mg/L for milk, mg/kg for beef)
F_{iA} = Element (stable) transfer coefficient that relates the daily intake by an animal to the concentration of i^{th} contaminant in an edible portion of the animal product (day/L for milk, day/kg for meat)
C_{if} = Concentration of i^{th} contaminant in forage (pCi/kg) (mg/kg)
Q_f = Consumption rate of contaminated forage by an animal (kg/day)
C_{iAw} = Concentration of i^{th} contaminant in livestock water (pCi/L) (mg/L)
Q_{Aw} = Consumption rate of contaminated water by an animal (L/day)

Site-specific data on radionuclides in milk, available in FEMP Environmental Monitoring Reports, will be used to supplement model predictions for current exposure scenarios.

In addition to intake from irrigated forage and water, cows may receive a significant intake from soil ingestion if the soil is also a source of contamination (Zach and Mayoh 1984). The following equation can be used to calculate the concentration in the animal product from soil ingestion (EPA 1989a):

$$C_{iA} = F_{iA}[(C_s)(Q_s)] \qquad (7\text{-}16)$$

where

C_s = Concentration of contaminant in soil (pCi/kg) (mg/kg)
Q_s = Consumption rate of soil by livestock (kg/day)

Once the concentration in the animal product is determined, human intake can be calculated using the following equations:

(radionuclides) $I_{iA} = (C_{iA})(IR)(ED)(FI)$ (7-17)
(chemicals) $I_{iA} = (C_{iA})(IR)(FI)(EF)(ED)/(BW)(AT)$ (7-18)

where

I_{iA} = Intake of chemical in animal product (pCi) (mg/kg-day)
C_{iA} = Concentration of i^{th} contaminant in the animal product (pCi/L for milk, pCi/kg for beef) (mg/L for milk, mg/kg for beef)
IR = Ingestion rate (L/yr for milk; kg/yr for beef) (L/day for milk; kg/day for beef)
FI = Fraction ingested from contaminated source (unitless)
EF = Exposure frequency (days/yr)
ED = Exposure duration (yr)
BW = Body weight (kg)
AT = Averaging time (days); for noncarcinogens, AT equals (ED)(350 days/yr); for carcinogens, AT equals (70-year lifetime)(365 days/yr)

7.2.1.6 Ingestion of Fish

Intake from ingestion of fish may require a one- or two-step process. If the concentration of a constituent in fish is unknown, it is necessary to determine the concentration in the fish based on the concentration in either the surface water or sediment (or both), for example:

$$C_F = (C_{sw})(BCF_F) \quad (7\text{-}19)$$

where

C_F = Concentration in the fish meat (pCi/kg) (mg/kg)
C_{sw} = Concentration in surface water (pCi/L) (mg/L)
BCF_F = Fish bioconcentration factor (pCi/kg fish per pCi/L) (mg/kg fish per mg/L)

Once the concentration in fish has been determined, or if measured concentrations in edible portions of fish are available, intake can be calculated as (EPA 1989a):

(radionuclides) $I_F = (C_F)(IR)(FI)(ED)$ (7-20)
(chemicals) $I_F = (C_F)(IR)(FI)(ED)(EF)/(BW)(AT)$ (7-21)

where

I_F = Intake from fish ingestion (pCi) (mg/kg-day)
C_F = Concentration in fish (pCi/kg) (mg/kg)
IR = Ingestion rate (kg/yr) (kg/day)
FI = Fraction ingested from contaminated source (unitless)

EF = Exposure frequency (day/yr)
ED = Exposure duration (yr)
BW = Body weight (kg)
AT = Averaging time (days); for noncarcinogens, AT equals (ED)(350 days/yr); for carcinogens, AT equals (70-year lifetime)(365 days/yr)

7.2.1.7 Dermal Contact with Soil or Water

For most metals, and hence most radionuclides at the FEMP, dermal absorption is not a significant pathway because penetration through the skin is minimal. However, it may be necessary to evaluate dermal absorption if organic constituents are found to contribute to potential risks at the site. The amount of a chemical taken into the body upon exposure via dermal contact is referred to as an absorbed dose and is calculated using the following equation (EPA 1989a):

$$AB_w = (C_w)(SA)(PC)(ET)(ED)(EF)/(BW)(AT) \qquad (7\text{-}22)$$

where

AB_w = Absorbed dose from contact with water (mg/kg-day)
C_w = Concentration in water (mg/L)
SA = Skin surface area available for contact (cm^2)
PC = Dermal permeability constant ($L/cm^2/hr$)
ET = Exposure time (hr/day)
ED = Exposure duration (yr)
EF = Exposure frequency (day/yr)
BW = Body weight (kg)
AT = Averaging time (days); for noncarcinogens, AT equals (ED)(350 days/yr); for carcinogens, AT equals (70-year lifetime)(365 days/yr)

Dermal absorption may also occur upon contact with contaminated soil and sediment and is calculated using the following equation (EPA 1989a):

$$AB_s = (C_s)(CF)(SA)(AF)(ABS)(ED)(EF)/(BW)(AT) \qquad (7\text{-}23)$$

where

AB_s = Absorbed dose from contact with soil (mg/kg-day)
C_s = Concentration in soil (mg/kg)
SA = Skin surface area available for contact (cm^2/event)
AF = Skin adherence factor (mg/cm^2)
ABS = Absorption factor (unitless)
CF = Conversion factor; (10^{-6} kg/mg)
ED = Exposure duration (yr)
EF = Exposure frequency (days/yr)

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BW = Body weight (kg)
AT = Averaging time (days); for noncarcinogens, AT equals (ED)(350 days/yr); for carcinogens, AT equals (70-year lifetime)(365 days/yr)

7.2.1.8 <u>Incidental Ingestion of Surface Water While Swimming</u>

Intake from incidental ingestion of surface water while swimming is quantified using the following equation (EPA 1989a):

$$\text{(radionuclides)} \quad I_{WS} = (C_{WS})(CR)(ET)(EF)(ED) \qquad (7\text{-}24)$$
$$\text{(chemicals)} \quad I_{WS} = (C_{WS})(CR)(ET)(EF)(ED)/(BW)(AT) \qquad (7\text{-}25)$$

where

I_{WS} = Intake from water while swimming (pCi) (mg/kg-day)
C_{WS} =Concentration in water (pCi/L) (mg/L)
CR = Contact rate (0.05 L/hr)
ET = Exposure time (hr/event)
EF = Exposure frequency (events/yr)
ED = Exposure duration (yr)
BW = Body weight (kg)
AT = Averaging time (days); for noncarcinogens, AT equals (ED)(350 days/yr); for carcinogens, AT equals (70-year lifetime)(365 days/yr)

7.2.1.9 <u>External Exposure</u>

The radiation dose equivalent resulting from exposure to direct penetrating radiation is calculated in the following manner:

$$DE = (DR)(EF)(ED)(MF)(SH) \qquad (7\text{-}26)$$

where

DE = Dose equivalent (mrem)
DR = Dose equivalent rate (mrem/day)
EF = Exposure frequency (days/yr)
ED = Exposure duration (yr)
MF = Modifying factor for hours spent outdoors; hours indoors; (unitless)
SH = Building shielding factor for dose equivalent rate reduction indoors (unitless)

7.2.2 <u>Intake and Exposure Model Parameter Values</u>

This tabulation of parameters and numerical parameter values has been established for use in the intake and exposure models. Parameter values are selected from a hierarchy of data sources. Parameter values will be obtained from site-specific data whenever possible. In the absence of site-specific data, parameter values recommended by EPA will be used. If these parameter values are not available from these sources, other sources will be used. Consistent use of parameters will be attempted for all models and scenarios unless deviations are clearly justified. The data sources in descending order of their position on the hierarchy are:

- Site-specific data (may include regional data)
- U.S. EPA Risk Assessment Guidance for Superfund, Volume I: Human Health Evaluation Manual, including supplemental guidance documents and suggested reference materials and services (e.g., EPA 1989a and EPA 1991c)
- U.S. EPA reports and other guidance documents, (e.g., EPA 1989f, EPA 1988c, EPA 1989b, EPA 1991d and Schaum 1991)
- National Academy of Sciences, BEIR IV (NAS 1988)
- National Academy of Sciences, BEIR V (NAS 1990)
- UNSCEAR Reports (UNSCEAR 1977, UNSCEAR 1982, UNSCEAR 1988)
- International Commission on Radiological Protection publications (e.g., ICRP 1975)
- Nuclear Regulatory Commission reports and guidance (e.g., Regulatory Guide 1.109 [NRC 1977])
- National Council on Radiation Protection and Measurements (NCRP) reports (e.g., NCRP 1984a; NCRP 1984b; NCRP 1984c; NCRP 1986)
- DOE publications (e.g., DOE 1989a; Baes et al. 1984)
- Other literature sources

The parameter values listed in this section are used in the exposure scenarios developed for the FEMP. Parameter values are identified with the parameter symbols used in the intake and exposure models listed in Section 7.2.1. Section 7.2.2.1 presents parameter values that describe human and animal receptors. Section 7.2.2.2 presents agricultural parameter values. Agricultural parameter values that are specific to southwest Ohio are used when available; default parameter values are used when site-specific data are not available. Section 7.2.2.3 presents chemical-specific parameter values used in intake and exposure models.

7.2.2.1 Human and Animal Descriptive Parameters

It is assumed in the RME scenario that a resident lives in the same home for a 70-year lifetime (EPA 1989a). The RME is considered as an adult exposure for most pathways. Exposures that are received only during childhood (e.g., sediment ingestion while playing in a creek) are addressed using a shortened exposure period and parameter values describing child exposure patterns. For evaluation of the nonstochastic health effects from chemical toxicity, an adult exposure scenario is generally used. However, in all cases risks to the most critically effected populations and age groups will be identified and presented. In addition, risks to different age groups can be combined to reflect composite exposures.

Human Physiological Parameters[a]

	Age (yrs)	Body Wt (kg)
Young Child	$0 < a < 6$	15
Child/Teen	$6 \leq a < 18$	43
Adult	$18 \leq a < 70$	70

[a] From EPA 1989f
[b] Extremity data from EPA 1989f will be used as necessary
[c] N/A - not available

Surface Area (m^2)			
Applicable Pathway(s): Body Part	Child < 6 yrs	Child/Teen 6-18 yrs	Adult over 18 yrs
Swimming, bathing: Total body	0.7[a]	1.33[b]	1.81[c]
Playing in creek:			
Forearms		0.078[d]	
Hands		0.057[d]	
Lower Legs		0.150[d]	
Feet		0.077[d]	
Dermal contact with soil during gardening, remediation activities:			
Forearms		0.078[d]	0.114[e]
Hands		0.057[d]	0.079

[a] Approximated from 50 percentile, ages 2-6; Table 2-4, EPA 1991d.
[b] Mean of 50 percentile values for ages 6-18; Table 2-4, EPA 1991d.
[c] Average adult (men and women); Table 2-3, EPA 1991d.
[d] Based on teen total body and a percentage of adult total body.
[e] Men only.

Exposure Duration (ED)

Reasonable maximum lifetime exposure	70 years[a]
Soil ingestion scenario	
	6 years as a young child (0<a<6)
	12 years as an older child (6≤a<18)
	52 years as an adult (18≤a<70)
Sediment ingestion scenario (child/teen, ages 6 to 18)	12 years[c]

[a] Agreement between DOE, Ohio EPA, and EPA on July 17, 1991
[b] (EPA 1989f)
[c] Assumption

Time Use Patterns (EF), (MF)[a]

Fraction of time spent indoors	0.5
Fraction spent sleeping	0.34
Fraction spent awake indoors	0.16
Fraction of time spent outdoors	0.5

[a] NCRP 1984a

Ingestion Rates of Home-Produced Foodstuffs (IR)

Consumption values reported by EPA (EPA 1989f) reflect results of the Nationwide Food Consumption Survey for 1978 (USDA 1980). The more recently published Nationwide Food Consumption Surveys for 1985 (USDA 1986a; USDA 1986b) reflect changing eating patterns in the United States, and thus are used in place of values reported by EPA (EPA 1989f). Data from the 1977 survey are presented in parentheses for comparison purposes. Data reported are mean values, except for drinking water and milk, which are maximum or worst-case values. Values for adult food consumption are obtained from supplemental guidance for human health evaluations (EPA 1991c) and account for the fraction of food obtained from a home-produced source.

Pathway	Infant	Child[a]	Adult[b]
Total veg. and fruits (g/day)[c]	-	303 (233)	122
Beef, pork, poultry (g/day)[c]	-	39 (46)	75
Fish and shellfish (g/day)[c]	-	5 (5)	54
Drinking water (L/day)	0.9[d,e]	1.4[d,e]	2.0[d,e,f]
Milk (L/day)[d,e]	0.9[d,e]	0.9[d,e]	0.30

[a] The values reported here for vegetable, fruit, beef, pork, poultry, and fish consumption are for children ages 1-5 (USDA 1986a).
[b] (EPA 1991c); assumed fraction home produced already included. The exposure for recreational consumption of locally caught fish is not added to exposures from other pathways, but is considered a pathway for a sensitive subpopulation.
[c] (USDA 1986a) and (USDA 1986b)
[d] (NRC 1977)
[e] (NCRP 1984a)
[f] (EPA 1989f)

<u>Fraction of Food Consumed from Source (FI)</u>

The following values are used to represent the percentage of a person's diet that comes from home-produced foodstuffs and site soils and sediment. Adult food consumption values presented already account for the percentage of an adult diet that comes from a home-produced source.

Item Ingested	Fraction Home-Produced
Vegetables	0.40[a]
Fruits	0.30[a]
Beef	0.75[a]
Dairy products	0.75[a]
Fish	0.75[b]
Drinking water	1.00[c]
Soil/Sediment	1.00[c]

[a] (EPA 1991c), 95th percentile values
[b] (EPA 1991c), "reasonable worst-case" value
[c] conservative assumption

<u>Human Soil and Sediment Ingestion</u>

Mass Ingested	Infant	Child/Teen	Adult	Lifetime Average
Exposure Duration (yr)[a]	6	12	52	70
Ingestion Rate (g/day)[b]	0.2	0.1	0.1	0.109[c]
Soil Ingestion Scenario				
Exposure Frequency (days/yr)[b]	350	350	350	350
Sediment Ingestion Scenario				
Exposure Frequency (days/yr)[d]		274		
Total Sediment Ingested (g)		329		

[a] EPA 1989f, reflecting risks to possible lifetime residence at nearby farms
[b] EPA 1991c
[c] Time-weighted average over 70 years
[d] Assumed

Age Adjusted Ingestion Rates

Receptor Group	Time-weighted Ingestion Rate (mg/d)	Total Soil Ingested (g)	Age Adjusted Ingestion Rate
Chemicals:			
Child, age 0<a<6	200	420	80 mg-yr/kg-day
Adult + teen, age 6≤a≤70	100	2240	91 mg-yr/kg-day
Adult, age 18≤a≤70	100	1820	74 mg-yr/kg-day
Human, to age 70	109	2660	171 mg-yr/kg-day
Sediment eater, age 6≤a<18	100	420	28 mg-yr/kg-day
Radionuclides:			
Sediment eater, age 6≤a<18	100	420	1200 mg-yr/day
Human, to age 70	109	2660	7600 mg-yr/day

Human Inhalation Rates (IR)

For continuous adult exposure situations in which specific activity patterns are not known, an adult inhalation rate of 20 m^3/day is used (EPA 1989a; EPA 1989f). For adult exposure situations in which the distribution of activity patterns is known, the following inhalation rates, and percentages at each activity level will be used:

Activity	Inhalation Rate (m^3/hr)	Percent of time at activity level[a]: Outdoor Average	Outdoor RME	Indoor Average	Indoor RME
Resting	0.5	28%	0%	48%	25%
Light	0.6	28%	0%	48%	60%
Moderate	2.1	37%	50%	3%	10%
Heavy	3.9	7%	50%	1%	5%

[a] EPA 1989f

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Animal Consumption Rates (Q_f, Q_{Aw}, Q_s)

The following parameters will be used to quantify the intake of contaminants in food and water by beef and milk cattle at or near the FEMP:

Animal	Q_f Feed or forage[a] (kg wet weight/day)	Q_{Aw} Water[a] (L/day)	Q_s Soil[b] (kg/day)
Milk cow	50	60	0.5 kg/day
(modified)[c]	25		
Beef cattle	50	50	0.5 kg/day
(modified)[c]	25		

[a] (NRC 1977)
[b] (Zach and Mayoh 1984)
[c] Modified assuming that pastureland is not irrigated due to the cost involved and based on data from the Bureau of Census (Bureau of Census 1989). Pasture forage is assumed to be supplemented with stored feed that was irrigated with contaminated water, and the animal diet consists of equal parts of pasture grass and stored feed totaling 50 kg/day wet weight.

7.2.2.2 Agricultural Parameters for Southwest Ohio

The growing season for feed corn in Hamilton County is 138 days (USDA 1970). Farms in the area have been known to use irrigation to supplement natural rain fall. Overhead sprinklers are the predominant form of irrigation equipment used. Typical irrigation requirements for feed corn in Hamilton County are about 10.6 inches/yr (0.081 L/m^2-hr) (USDA 1970). Additional parameters are listed in Table 7-2.

7.2.2.3 Chemical-Specific Parameters

Other Radionuclides, Nonradioactive Inorganic Transfer Factors (F_{iA})

Transfer coefficients for nonuranium radioelements and nonradioactive metals are taken from Baes et al. (1984), Till and Meyer (1983), and DOE (1989a). The radiological properties of atoms do not effect their elemental transfer in the environment.

The following are soil-to-plant concentration factors for edible plants consumed by man used in intake models in the absence of site-specific information. These factors are the ratios of the dry-weight concentration of an element in the reproductive portions of the plant to the dry-weight concentration of the element in soil. Reproductive portions of the plant include grain kernels, fruits, and tubers. These portions are most indicative of the plant foods consumed by man.

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TABLE 7-2
SUMMARY OF PARAMETERS FOR VEGETABLE/FORAGE UPTAKE MODELS

Parameter:	Value:	Units:	Reference:
Irrigation rate (d_i):	0.081	$L/m^2/hr$	USDA 1970
Fraction of deposited dust retained on crops (r_d):	0.25	unitless	NRC 1977
Fraction of irrigation deposits retained on crops (r_w):	0.20	unitless	NRC 1977
Removal rate by weathering (λ_{Ei}):	0.0021	hr^{-1}	NRC 1977
Growing season for crops (t_{ec}):	1440	hr	NRC 1977
Growing season for forage (t_{eg}):	720	hr	NRC 1977
Growing season for feed (t_{ef})	2160	hr	NRC 1977
Agricultural yield of food crops (Y):	1.5	kg/m^2	USDA 1979
Agricultural yield of forage (Y):	0.8	kg/m^2	USDA 1979
Fraction of year plants are downwind (f_d):	LD[a]	unitless	-
Fraction of year plants are irrigated (f_w):	0.38	unitless	NRC 1977
Period soil is exposed to contaminated water (t_{bw}):	LD	hr	-
Period soil is exposed to airborne emissions (t_{bd}):	LD	hr	-
Effective surface density (ρ):	150	kg/m^2	b
Delay between harvest and consumption of vegetables (t_{hv}):	24	hr	NRC 1977
Delay between harvest and consumption of fruit (t_{hc}):	720	hr	Assumed
Delay between harvest and consumption of feed (t_{hf}):	2160	hr	NRC 1977
Delay between harvest and consumption of forage (t_{hg}):	0	hr	NRC 1977
Delay between milking and consumption:	48	hr	NRC 1977
Delay between slaughter and consumption:	480	hr	NRC 1977

[a] Location dependent

[b] Corresponds to a density of 1.5 g/cm^3 and a depth of 10 cm. Moist bulk densities of surface soil range from 1.4 to 1.55 g/cm^3 at the FEMP (USDA 1982).

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Element	Concentration Ratio[a] (F_{iA}, $B_{iv(2)}$)
Sr	2.5×10^{-1}
Tc	1.5×10^{0}
Pb	9.0×10^{-3}
Po	4.0×10^{-4}
Ac	3.5×10^{-4}
Th	8.5×10^{-5}
Pa	2.5×10^{-4}
Np	1.0×10^{-2}
Pu	4.5×10^{-5}
Ra	1.5×10^{-3}
Cs	3.0×10^{-2}
Ru	2.0×10^{-2}

[a] Baes et al. 1984

Organic Transfer Factors (F_{iA})

Transfer coefficients for organic chemicals are taken from Travis and Arms (1988). If a transfer coefficient is not readily available, the following regression equations based on the relationship between transfer and the octanol-water partition coefficient (K_{ow}) are used to estimate transfer coefficients (Travis and Arms 1988):

$B_{iv(2)}$ (vegetables) $\quad \log B_{iv} = 1.588 - 0.578 \log K_{ow}$ (7-27)

F_{iA} (milk) $\quad \log F_{iA} = -8.10 + \log K_{ow}$ (7-28)

F_{iA} (beef) $\quad \log F_{iA} = -7.6 + \log K_{ow}$ (7-29)

Chemical-specific K_{ow} values are available from several sources. The major source used for K_{ow} values is Hansch and Leo (1979).

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Skin Permeability Constant (PC)

Chemical-specific skin permeability constants (PC) are obtained from EPA (1991d and Schaum 1991) for specific volatiles and semivolatiles. The following PCs will be used (Schaum 1991):

Compound	Permeability Constant (cm/hr)
vinyl chloride	0.007
1,2-dichloroethylene	0.01
chlorobenzene	0.04
xylenes	0.08
1,2-dichlorobenzene	0.06
4-methyl phenol	0.06
naphthalene	0.07
pentachlorophenol	0.6
fluoranthene	0.4

For other organics, the following equation, which correlates PC with the octanol-water partition coefficient (K_{ow}) and molecular weight (MW), will be used:

$$\log K_p = -2.73 + 0.71 \log K_{ow} - 0.0061(MW) \qquad (7\text{-}30)$$

For specific inorganics, the following PCs will be used:

Compound	Permeability Constant (cm/hr)
cobalt	0.0004
lead	0.000004
silver	0.0006
zinc	0.0006

For other inorganics, assume 1×10^{-3} cm/hr.

Dermal Absorption Values (ABS)

As specified by EPA (1991d) and Schaum (1991), dermal contact with soil and waste will be quantitatively evaluated for dioxins, furans, PCBs, DDT, and cadmium. Volatile compounds are not quantitatively evaluated because it may be assumed that they do not contribute significant risks via dermal contact with soil. For other organics, dermal absorption will be assessed (either qualitatively or quantitatively) using dermal absorption values from the literature. Chemical-specific dermal absorption values will be taken from Schaum (1991) for the following chemicals:

dioxins and furans	10%
PCBs	10%
DDT, DDD, DDE	30%
cadmium	0.1%

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Radionuclide-Specific Activities

The following specific activities are used to convert from activity to mass:

Radionuclide	Specific Activity[a] (pCi/μg)
Actinium-227	7.24E+07
Cesium-137	8.65E+07
Neptunium-237	7.05E+02
Plutonium-238	1.71E+07
Plutonium-239	6.21E+04
Protactinium-231	4.72E+04
Lead-210	7.64E+07
Radium-224	1.59E+11
Radium-226	9.89E+05
Radium-228	2.72E+08
Strontium-90	1.37E+08
Technetium-99	1.70E+04
Thorium-228	8.20E+08
Thorium-230	2.06E+04
Thorium-232	1.10E-01
Uranium-234	6.22E+03
Uranium-235	2.16E+00
Uranium-238	3.36E-01

[a] DHEW 1970

Conversion from Total Activity (pCi) to Mass (μg) for Uranium:

Total mass of 1 μg uranium = 0.66 pCi, or
Total activity of 1 pCi uranium = 1.5 μg[a]

[a] NCRP 1984c; this uranium conversion factor between total activity and total mass incorporates the assumption that the naturally occurring uranium isotopes (uranium-234, uranium-235, uranium-238) are present in their naturally occurring percent mass abundances (0.0055% uranium-234, 0.72% uranium-235, 99.27% uranium-238). Therefore, 1 μg total uranium converts to approximately 0.66 pCi total uranium activity, of which approximately half is uranium-234 activity and half is uranium-238 activity.

Radiation Shielding Factor (SH)

An indoor shielding factor of 0.5 will be used as suggested by the NRC (1977).

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7.2.3 Quantitative Exposure Assessment Results

Intake model equations for radionuclides and for hazardous chemicals are presented. In general, intake for radionuclides and chemicals is calculated in a similar manner with the following exceptions:

- The unit for radionuclide intake is pCi, while the unit for chemical intake is generally mg.
- Radionuclide intakes are expressed as total intakes, while chemical intakes are expressed as daily intakes per unit body weight.

Quantitative intake estimates usually constitute the end result of the exposure assessment process. In the RI and FS risk assessments, these intake estimates are used in conjunction with contaminant toxicity data to estimate the risks associated with the RME for each pathway.

7.3 RADIATION DOSE ASSESSMENT

Radiation doses resulting from the potential exposures of a receptor to radionuclides will be calculated as part of this risk assessment. Note that the term "dose" has a different meaning for radionuclides than that for chemicals. Radiation dose is defined as the energy imparted to a unit mass of tissue; the dose unit is usually joule per kilogram of tissue, whereas the chemical dose can be defined as the mass penetrating into an organism; the dose unit is usually milligram per kilogram. It has been long recognized that the absorbed radiation dose needed to achieve a given level of biological damage varies for different types of radiation (alpha-particles, beta-particles, gamma rays, or neutrons). For radiation protection purposes, it is desirable to use a quantity for all types of ionizing radiation, that correlates to the biological effect on a common scale. This quantity is the dose equivalent and has units of rem or millirem (mrem). The dose equivalent is defined as the product of the absorbed dose and a quality factor, which depends on the relative biological effectiveness of the radiation at the point of interest in tissue. A quality factor of unity is used when calculating the dose equivalent for penetrating radiation (e.g., gamma rays).

Dose assessment is necessary for two reasons. First, calculated doses are required for comparison to ARARs. Second, most of the source geometries at the FEMP preclude the use of EPA external gamma slope factors, which were only calculated for one geometry (surface soil lying in a plain). The geometry used by EPA (1991a) is a flat source, 10 cm thick, with a surface density of 143 kg/m^2, which is representative of contaminated surface soil. Another method must therefore be used to estimate the risks from sources with other geometries.

Microshield 3.0, described in Section 6.5, will be used to calculate exposure rates from external sources at the FEMP. Doses resulting from these exposure rates will be calculated using

Equation 7-26. These doses will be used in conjunction with a dose to risk conversion factor (Section 9.2.2.2) to estimate risks from external radiation from radiological sources other than surface soil.

7.4 ECOLOGICAL EXPOSURE ASSESSMENT

This section describes the methods used to estimate the exposures to ecological receptors from exposure to constituents of concern at the FEMP. Current concentrations of constituents will be estimated from RI/FS and environmental monitoring data. Future concentrations will be estimated by fate and transport modeling.

7.4.1 Plants

Concentrations of radionuclides in plants at the FEMP were measured in 1987 and 1988 as part of the RI/FS. These concentrations, which were measured when the FEMP was still in production, may include contributions from air deposition of stack emissions and therefore may not be representative of present conditions. However, these concentrations should represent the upper bound for radionuclide concentrations in vegetation at the FEMP. A lower bound will be estimated from soil radionuclide data, using soil-to-plant transfer factors provided by Baes et al. (1984) (Table 7-3) and assuming that the only mechanism for radionuclide accumulation in plants is uptake from soil and aerial deposition onto the plants.

Because RI/FS data on the concentrations of inorganic and organic constituents in FEMP vegetation are limited to 10 grass samples, additional estimates will be made of the maximum concentrations of these constituents in a generic plant growing in FEMP soil. Soil-to-plant (aboveground vegetative portion) transfer factors for organic constituents obtained from Baes et al. (1984) are presented in Table 7-4. Soil-to-plant transfer coefficients for organic compounds of potential concern will be estimated from K_{ow} values listed in Table 6-4, as described by the footnote at the bottom of Table 7-4.

Calculated transfer factors for organic constituents of potential concern identified to date are presented in Table 7-4. The transfer factors used for both metals and organics are conservative estimates and do not consider such factors as the bioavailability of a chemical in soil, the biodegradation rate of a compound in soil, or metabolic transformations of compounds in plants.

The maximum concentration of each constituent of potential concern measured in FEMP surface soil (composite soil data will be used when surface soil data are unavailable) will be used as the

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TABLE 7-3
SOIL-TO-PLANT AND PLANT-TO-BEEF TRANSFER COEFFICIENTS USED FOR RADIONUCLIDES AND INORGANIC CHEMICALS OF POTENTIAL CONCERN IN FEMP SOILS

	Transfer Coefficient	
Chemical	Soil-to-Plant[a] ($B_{iv(2)}$)	Plant-to-Beef (B_{ib})
Radioelements		
Cesium	0.080	0.020
Neptunium	0.10	5.5×10^{-5}
Plutonium	0.00045	5.0×10^{-7}
Radium	0.015	2.5×10^{-4}
Strontium	2.5	3.0×10^{-4}
Thorium	0.00085	6.0×10^{-6}
Uranium	0.0085	2.0×10^{-4}
Inorganic Chemicals		
Arsenic	0.04	2.0×10^{-3}
Barium	0.15	1.5×10^{-4}
Beryllium	0.010	1.0×10^{-3}
Cadmium	0.55	5.5×10^{-3}
Chromium	0.0075	5.5×10^{-3}
Cobalt	0.020	0.020
Copper	0.40	0.010
Lead	0.045	3.0×10^{-4}
Magnesium	1.0	5.0×10^{-3}
Manganese	0.25	4.0×10^{-4}
Mercury	0.90	0.25
Nickel	0.060	6.0×10^{-3}
Selenium	0.025	0.015
Thallium	0.004	0.040
Vanadium	0.0055	2.5×10^{-3}
Zinc	1.5	0.10

[a] Soil-to-plant elemental transfer factor for vegetative portions of food and feed plants. It assumes dry plant and soil weights (Baes et al. 1984).

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TABLE 7-4
SOIL-TO-PLANT AND PLANT-TO-BEEF TRANSFER COEFFICIENTS USED FOR ORGANIC COMPOUNDS OF POTENTIAL CONCERN IN FEMP SOILS

	Transfer Coefficients	
Compound	Soil-to-Plant[a] ($B_{iv(2)}$)	Plant-to-Beef[b] (B_{ib})
Polycyclic aromatic hydrocarbons		
Acenapthene	0.16	3.0×10^{-4}
Anthracene	0.104	7.0×10^{-4}
Benzo(a)anthracene	0.022	0.010
Benzo(a)pyrene	0.013	0.0275
Benzo(b)fluoranthene	6.2×10^{-3}	0.093
Benzo(g,h,i)perylene	2.6×10^{-3}	0.427
Benzo(k)fluoranthene	4.3×10^{-3}	0.178
Chrysene	0.022	0.010
Dibenzo(a,h)anthracene	0.017	0.0155
Fluoranthene	0.032	5.4×10^{-3}
Fluorene	0.149	4.0×10^{-4}
Indeno(1,2-cd)pyrene	1.4×10^{-3}	1.15
Naphthalene	0.479	1.0×10^{-4}
Phenanthrene	0.102	7.0×10^{-4}
Pyrene	0.033	0.0052
Monocyclic Aromatics		
Benzene	2.27	3.4×10^{-6}
Benzoic Acid	3.21	1.9×10^{-6}
Chlorobenzene	0.88	1.7×10^{-5}
2, 4-Dimethylphenol	1.39	7.9×10^{-6}
Ethyl benzene	0.585	3.6×10^{-5}
2-Methylphenol	2.89	2.2×10^{-6}
4-Methylphenol	2.93	2.2×10^{-6}
Pentachlorophenol	0.046	2.9×10^{-3}
Phenol	5.55	7.2×10^{-7}
Toluene	1.02	1.35×10^{-5}
Xylenes, total	0.585	3.55×10^{-5}

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TABLE 7-4

(Continued)

	Transfer Coefficient	
	Soil-to-Plant[a]	Plant-to-Beef[b]
Compound	($B_{iv(2)}$)	(B_{ib})
Phthalate esters		
Bis(2-ethylhexyl)phthalate	0.043	3.2×10^{-3}
Butyl benzyl phthalate	0.056	2.0×10^{-3}
Di-n-butyl phthalate	0.072	1.3×10^{-3}
Di-n-octyl phthalate	2.0×10^{-4}	39.8
Polychlorinated biphenyls		
Aroclor 1016	0.11	6.0×10^{-4}
Aroclor 1242	0.16	3.0×10^{-4}
Aroclor 1248	0.022	0.01
Aroclor 1254	7.1×10^{-3}	0.074
Aroclor 1260	0.011	0.032
Halogenated aliphatic hydrocarbons		
Chloroform	2.81	2.3×10^{-6}
1,1-Dichloroethane	3.58	1.55×10^{-6}
1,1-Dichloroethene	5.40	7.6×10^{-7}
1,2-Dichloroethene	2.5	2.9×10^{-6}
Methylene chloride	7.34	4.5×10^{-7}
1,1,2,2-Tetrachloroethane	0.42	1.0×10^{-4}
Tetrachloroethene	0.42	1.0×10^{-4}
1,1,1-Trichloroethane	1.41	7.8×10^{-6}
Trichloroethene	1.84	4.9×10^{-6}
1,1,2-Trichloro-1,2,2-		
Vinyl chloride	6.17	6.0×10^{-7}
Nonhalogenated aliphatic hydrocarbons		
Acetone	53.3	1.45×10^{-8}
2-Butanone	26.3	4.9×10^{-8}
4-Methyl-2-pentanone	7.95	3.9×10^{-7}

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TABLE 7-4
(Continued)

Compound	Transfer Coefficient: Soil-to-Plant[a] ($B_{iv(2)}$)	Transfer Coefficient: Plant-to-Beef[b] (B_{ib})
Pesticides		
Beta-BHC	0.246	2.0×10^{-4}
Chlordane	0.013	0.025
4,4-DDT	0.018	0.0145
Malathion	0.827	1.95×10^{-5}
Methyl parathion		
Miscellaneous Compounds		
Carbon disulfide	2.19 - 3.35	1.7×10^{-6} to
N-Nitrosodiphenylamine	1.27	9.3×10^{-6}
3,3-Dichlorobenzidine	0.70	2.6×10^{-5}

[a] Soil-to-plant transfer coefficients from Travis and Arms (1988); based on dry plant weight and dry soil weight [log Biotransfer Factor = 1.588 - 0.578 log K_{ow}]

[b] Soil-to-beef transfer coefficients from Travis and Arms (1988); assumes meat is 25% fat [log Biotransfer Factor = -7.6 + log K_{ow}] (Travis and Arms 1988)

exposure concentration in each case. Concentrations in the aboveground vegetative part of plants will be estimated using the following equation (Baes et al. 1984):

$$C_{iv} = (C_s)(B_{iv(2)}) \qquad (7\text{-}31)$$

where

C_{iv} = Concentration of the i^{th} contaminant in vegetation (mg/kg dry wt)
C_s = Maximum concentration in soil (mg/kg dry wt)
$B_{iv(2)}$ = Soil to plant transfer factor of the i^{th} contaminant (mg/kg dry wt plant per mg/kg dry wt soil)

7.4.2 Terrestrial Animals

7.4.2.1 Intake of Radioactive and Nonradioactive Constituents

The maximum concentrations of constituents of concern in selected terrestrial indicator species will be estimated as described in the following paragraphs. The selection of terrestrial indicator species was based on species abundance on the FEMP, trophic level position, and habitat requirements. Terrestrial indicator species for the FEMP include the white-tailed deer (Odocoileus virginianus), white-footed mouse (Peromyscus leucopus), raccoon (Procyon lotor), muskrat (Ondatra zibethica), American robin (Turdus migratorius), red fox (Vulpes vulpes), and red-tailed hawk (Buteo jamaccnsis) (Facemire et al. 1990). Exposure pathways of indicator species to FEMP contaminants include the following:

- Ingestion of contaminated soil, vegetation and water, and exposure to external radiation by white-tailed deer
- Ingestion of contaminated vegetation, insects/earthworms and water, and exposure to external radiation by white-footed mice
- Ingestion of contaminated fruits, fish and water, and exposure to external radiation by raccoons
- Ingestion of contaminated wetland vegetation and water, and exposure to external radiation by muskrats
- Ingestion of contaminated fruits, earthworms and water, and exposure to external radiation by American robins.

- Ingestion of contaminated white-footed mice or white-tailed deer, fruits and water, and exposure to external radiation by red fox

- Ingestion of contaminated white-footed mice and water and exposure to external radiation by red-tailed hawk.

This modeling will be supplemented by RI/FS data on concentrations of radioactive and nonradioactive constituents in terrestrial animals at the FEMP. Nine samples were analyzed for radioactive constituents and four for organic and inorganic constituents.

Intake of constituents in vegetation by herbivores will be estimated using an equation adapted from EPA (1989a):

$$I_{iv} = (C_{iv})(IR_{iv})(FI)(EF)(ED)/(BW)(AT) \qquad (7\text{-}32)$$

where

I_{iv} = Intake of the i^{th} contaminant in vegetation (mg/kg-day)
C_{iv} = Concentration of the i^{th} contaminant in vegetation (mg/kg)
IR_{iv} = Ingestion rate (kg/day)
FI = Fraction ingested from contaminated source (unitless)
EF = Exposure frequency (days/yr)
ED = Exposure duration (yr)
BW = Body weight (kg)
AT = Averaging time, (ED)(350 days/yr)

Species-specific values for parameters such as ingestion rate and body weight will be developed as part of the ecological risk assessment.

In order to evaluate the potential exposure of resident red fox and red-tailed hawk to FEMP contaminants, estimates will be made of the concentrations of metals and organic compounds in the muscle tissues of a prey species. Concentrations of metals and organics in muscle tissue of white-footed mice will be calculated using plant-to-beef transfer factors developed for cows. The same procedure will be used for estimating contaminant uptake by white-tailed deer.

Plant-to-muscle transfer factors will be used instead of plant-to-whole animal transfer factors, due to the absence of such values from the literature. Use of plant-to-muscle transfer factors may underestimate the concentration of a contaminant in a prey species for some constituents that can be biomagnified through food chains and which concentrate in specific tissue (e.g., chlorinated organics in fat, lead and strontium in bone, and mercury in kidney and liver). Because of the lack of plant-to-whole body transfer factors and the absence of data on the amount of fat in FEMP animals, plant-to-beef transfer factors will be used along with assumptions that (1) the fat content in white-footed mice and white-tailed deer is minimal; (2) if bones of the prey species are ingested, most of the ingested bone will not be digested; and (3) concentrations of metals in a whole deer or white-footed mouse are expected to be similar to that in muscle. This is supported by data on omnivorous rodents in which whole body metal concentrations were within one order of magnitude of those in muscle (dry weight basis), as reported for cadmium (<1 to 2.25), lead (0.4 to 6.5), and zinc (1.3 to 1.7) (Talmage and Walton 1991). Whole body-to-muscle ratios for mercury in wild mammals were not found in the literature. However, comparisons of mercury in kidney to that in muscle indicate concentrations in the kidneys of omnivorous rodents of 0.5 to 2 times the concentration in muscle (dry weight basis) (Talmage and Walton 1991). Because mercury concentrates in kidney and liver tissues, this ratio is expected to be greater than the whole body-to-muscle concentration ratio. With these assumptions in mind, metal and radionuclide transfer factors for plant-to-beef were obtained from Baes et al. (1984) and are presented in Table 7-2. In addition, transfer factors for organic compounds were estimated using an equation derived by Travis and Arms (1988) and are presented in Table 7-3.

The concentration of a chemical in muscle will be estimated using the following equation:

$$C_{iA} = B_{ib}(C_{iv})(IR_{iv}) \qquad (7\text{-}33)$$

where

C_{iA} = Concentration of i^{th} contaminant in muscle (mg/kg)
B_{ib} = Plant-to-beef transfer factor (day/kg)
C_{iv} = Concentration of i^{th} contaminant in vegetation (mg/kg)
IR_{iv} = Ingestion rate of vegetation by animal (kg/day)

Parameters used in estimating intake by herbivores and omnivores include the concentration in vegetation. Concentrations in vegetation used in the intake calculations will be those estimated using the maximum soil concentration determined for the FEMP and the respective soil-to-plant transfer factor for a given chemical, as described previously.

Each of the equations used for herbivores can be modified for carnivores by substituting the concentration in herbivore muscle for vegetation. As a default value, the muscle-to-muscle transfer coefficient can be assumed to be one.

Exposure to soil constituents following direct ingestion of soil by wildlife will be evaluated by estimating intake in the same manner as described previously for intake of vegetation by an herbivore. Species-specific parameters associated with soil intake, such as ingestion rate, are currently under review. A default value of one will be assumed for the soil-to-muscle transfer coefficient. Ingestion of earthworms will be the primary route of exposure evaluated for the American robin. A default value of unity (1) will be assumed for the soil-to-earthworm transfer coefficient, due to the lack of soil-to-earthworm transfer coefficients in the literature.

In the event that more than one pathway is evaluated for a given indicator species, intake across all pathways will be summed to obtain a total intake value. For instance, uptake of a contaminated soil by white-tailed deer will be estimated by adding the intake via ingestion of vegetation and soil.

7.4.2.2 Radiation Doses to Terrestrial Animals

External exposures for animals will calculated in the same manner as those for humans (Section 6.4 and Section 7.3). Internal radiation absorbed doses (rad) (dose equivalent is defined only for humans) to terrestrial animals will be estimated from measured or estimated tissue radionuclide concentrations, assuming a uniform distribution in the organism, using the following equation:

$$\text{Calculated dose (rad/y)} = 0.01867(A)(C_{iA}) \qquad (7\text{-}34)$$

where

0.01867	=	Constant (rad y^{-1} pCi^{-1} g MeV^{-1} disintegration)
A	=	Mean energy of decay (MeV per disintegration)
C_{iA}	=	Radionuclide concentration in the organism (pCi per g dry weight)

The constant 0.01867 is derived in the following manner:

$$0.01867 = (A)(B)(C)(D)(E)(F)(G) \qquad (7\text{-}35)$$

where

A	=	1 Ci/10^{12} pCi
B	=	3.7 x 10^{10} disintegrations/Ci-sec
C	=	3600 sec/hr
D	=	8760 hr/yr
E	=	10^6 eV/MeV
F	=	1.6 x 10^{-12} erg/eV
G	=	1 rad-g/100 ergs

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For example, the energy of decay of uranium-234 is 4.8 MeV per decay and the energy of decay of uranium-238 is 4.2 MeV per decay (Kocher 1981). If the two isotopes are present in equal isotopic abundance in an organism, the average energy of 4.5 MeV per decay can be substituted in the equation, and the conversion factor is:

$$\text{Calculated dose (rad/y)} = 0.084(C_{iA}) \qquad (7\text{-}36)$$

or 84 mrad per year for each pCi uranium per gram dry weight. Similar calculations can be made for other radionuclides, substituting the appropriate energy of decay.

The radiation dose to a muskrat exposed to surface waters at the FEMP via water ingestion, food chain uptake, and direct exposure will also be estimated from surface water radionuclide concentrations using the constants provided by Killough and McKay (1976) (Table 7-5). This will assist in assessing radiological risks associated with links between the terrestrial and aquatic food chains.

7.4.3 Aquatic Organisms

Radionuclide concentrations in fish and macroinvertebrates from the Great Miami River and Paddys Run have been measured as part of the RI/FS. In addition, radionuclide concentrations in fish collected from the Great Miami River are reported in the annual Environmental Monitoring Reports compiled by WMCO (WMCO 1990). Radiation doses to fish and macroinvertebrates in the Great Miami River and Paddys Run will be estimated from these reported concentrations as described above for terrestrial animals. Radiation doses to aquatic organisms in the Great Miami River, Paddys Run, and on-site drainages will also be estimated from concentrations of radionuclides in surface water using the constants provided by Killough and McKay (1976) (Table 7-5).

Exposure of aquatic organisms to nonradioactive constituents of concern will be estimated from RI/FS surface water data on nonradioactive chemicals, assuming constant exposure. Future concentrations of nonradioactive constituents in surface waters will be estimated as described in Section 6.2. Characterization of risks to aquatic organisms as a result of exposure to radioactive and nonradioactive constituents is described in Section 9.0.

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TABLE 7-5

INTERNAL RADIATION DOSES (MRAD/Y) TO FRESH-WATER BIOTA EXPOSED TO 1.0 pCi/L[a]

	Receptor			
Radionuclide	Aquatic Plants	Invertebrates	Fish	Muskrat
Cesium-137	0.88	1.1	4.4	6.2
Radium-226	5,100	510	100	22,000
Strontium-90	10	2.1	0.1	44
Thorium-228	6,500	2,200	130	9.7
Thorium-230	1,300	450	27	1.9
Uranium-234	920	92	9.2	1.3
Uranium-235	860	86	8.6	1.2
Uranium-236	880	88	8.8	1.3
Uranium-238	800	8.0	8.0	1.2

[a] Adapted from Killough and McKay (1976)

8.0 TOXICITY ASSESSMENT

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A toxicity assessment consists of two stages:

- Toxicological evaluation
- Dose-response assessment

The first step in the toxicity assessment, the toxicological evaluation, is a qualitative evaluation of the scientific data to determine the nature and severity of the toxic properties associated with the radionuclides and chemicals of potential concern. The toxicological evaluation involves a critical review and interpretation of toxicity data from epidemiological, clinical, animal, and in vitro studies.

Once the potential adverse effects of a constituent have been characterized, the next step is a quantitative estimation of the amount of exposure to a constituent that may result in an adverse effect. This defines the relationship between the dose received by a constituent and the incidence of the adverse effect.

For noncarcinogens, it is assumed that a dose exists below which no adverse health effects will be seen (i.e., a threshold dose). For carcinogens, it is assumed that no threshold exists, and that any dose may result in a cancer. The probability of cancer development is described by the slope of the dose response curve. The following sections describe the information and sources of information that will be used to perform the toxicity assessment.

8.1 TOXICITY INFORMATION FOR NONCARCINOGENIC EFFECTS

Information on the toxic effects of noncarcinogens will be summarized both qualitatively and quantitatively. Qualitative toxicity information for noncarcinogenic effects will include information on general uses of the constituent, the critical studies used as a basis for the toxicity value, toxic effects resulting from acute and chronic exposure, critical toxic effect observed or target organ effected, and the absorption efficiency.

As an example, consider the element uranium, which is a major concern in the environment surrounding the FEMP. It is both chemically toxic and radioactive. Whether the chemical toxicity or radiotoxicity of uranium dominates in a given exposure scenario depends on the chemical form and the degree of isotopic enrichment. The physical particle size of the compound also becomes important when dealing with inhalation exposures.

The target organ for chemical toxicity of uranium is the kidney (Leggett 1989). In mammalian systems, uranium quickly reacts to form the uranyl ion. The uranyl ion forms stable complexes with the bicarbonate ions in the systemic circulation. However, at the kidney, where a substantial drop in pH occurs, the uranyl-bicarbonate complex dissociates. The uranyl ion binds to the kidney tissue, resulting in cellular necrosis (Leggett 1989).

The toxic effects of uranium will be addressed in detail in the risk assessments for the FEMP. The dose-response studies used to develop the uranium reference dose will be documented.

Quantitative information will be provided for each chemical toxicant of concern in the form of a table that will include the following information:

- Reference dose (RfD)
- Source of the RfD
- Critical effects on target organs
- Uncertainty factor used to develop the RfD

The two sources that will be used to identify RfD values are the IRIS database (EPA 1991b) and the most current edition of HEAST (EPA 1991a).

If relevant EPA-derived RfDs do not exist for constituents of concern, appropriate values will be derived. Justification will be provided for any derived values. Justification for any route-to-route extrapolation of an RfD or qualitative analysis of a constituent will be documented in this section. If lead is found to be of concern at the site, its toxicity will be evaluated with the EPA Uptake/Biokinetic Model (EPA 1990b).

8.2 TOXICITY INFORMATION FOR CARCINOGENIC EFFECTS

As with chemical toxicants, the health effects from carcinogens will be described with both a qualitative information summary and quantitative information, provided in tabular form. Qualitative information will include such information as principal effects, primary routes of exposure that result in adverse effects, and absorption rates.

As noted in the EPA report, Risk Assessment Guidance for Superfund Volume 1 Human Health Evaluation Manual (Part A) (EPA 1989a), fundamental differences exist between radionuclides and chemicals with respect to toxicity assessments. The principal adverse biological effects associated with radiation exposures from radioactive materials in the environment are carcinogenicity, mutagenicity, and teratogenicity (EPA 1989a). Of these, carcinogenicity is the limiting effect at low levels of radiation dose (environmental levels). The incidence-to-fatality ratio for radiogenic cancers is approximately two-to-one, when averaged over all cancer types (EPA 1989a). Data presented in HEAST (EPA 1991a) present the relationship between cancer incidence and exposure to radioactive materials.

The critical organ for the radiocarcinogenic effects of soluble forms of uranium is bone. For insoluble forms, the lung is the critical organ. The uranium isotopes of concern (U-234, U-235, and U-238) are all alpha particle emitters. Because epidemiological studies of uranium exposures generally have not been completed, information on radiation effects is based on animal studies and tumor rates from human populations exposed to other alpha-emitters. The most likely effect from exposure to soluble uranium compounds is an increase in bone sarcomas, while the most likely effect of insoluble forms of uranium is an increase in lung cancer.

Potential toxic effects of each radionuclide and chemical contaminant of concern at the site (or operable unit) will be discussed in the risk assessments. Results of the toxicity assessment will be summarized in tabular form to include the following information:

- Cancer slope factor (SF) by intake or exposure route
- Weight of evidence classification
- Type of cancer
- Basis for the SF

As with reference doses, quantitative toxicity information for radionuclides and chemicals will be obtained from IRIS and HEAST. The following exceptions are noted. Polycyclic Aromatic Hydrocarbons (PAH), for which no toxicity data are available, will be evaluated using benzo(a)pyrene toxicity data. Risks from exposure to penetrating radiations from sources other than radioactive materials in soil will be evaluated using a dose-based risk coefficient, because there is no conversion factor (slope factor) in HEAST for this exposure pathway. A risk coefficient of 6.2×10^{-7} $mrem^{-1}$ will be used for exposure to penetrating radiations from sources other than soil. This risk coefficient is taken from background information for the NESHAPS (EPA 1989b) and represents the currently accepted risk coefficient for estimating cancer incidence due to exposure to penetrating radiation. Uncertainties associated with the use of this coefficient will be presented in the risk assessments.

8.3 TOXICITY INFORMATION FOR ECOLOGICAL EFFECTS

Toxicity information for ecological effects will consist of No Observable Effects Concentrations (NOEC) and Lowest Observable Effects Concentrations (LOEC) for radionuclides and chemicals of potential concern and descriptions of the effects used to determine NOECs and LOECs. This information will be drawn from EPA Ambient Water Quality Criteria for the protection of aquatic life (EPA 1986a), Ohio Water Quality Standards (OEPA 1990b), and the literature. An additional reference that will be used is Effects of Radiation on Aquatic Organisms and Radiobiological Methodologies for Effects Assessment (EPA 1986b). Toxicity information for effects on terrestrial organisms will also rely on radioecology studies in the literature, U.S. Fish and Wildlife Service Studies (e.g., Eisler 1985), and the animal studies that support the HEAST and IRIS databases (EPA 1991a, EPA 1991b). More specifically, toxicity of chemicals to terrestrial species will be evaluated by comparisons of chemical-specific intake values to NOEC values. As a screening tool, NOEC and LOEC values presented in the IRIS database (EPA 1991b) will be used for mammals. Uncertainty factors will be applied to the animal toxicity data to correct for differences between species, to modify LOEC values to NOEC values, and to adjust data obtained through short-term studies to those which would be expected in long-term studies. Literature obtained avian toxicity values will be used for the robin. LD_{50} values will be adjusted with uncertainty factors to obtain an estimated NOEC. In the absence of avian toxicity data, available mammalian data will be substituted and appropriate uncertainty factors used. Uncertainty factors used to modify toxicity values will include:

- Short-term (<30 days)(Newell et al. 1987) effect levels will be multiplied by 0.1 to estimate chronic, long-term effects.

- LOECs will be converted to NOECs by multiplying the effect concentration by 0.2 (Newell et al. 1987).

- LD_{50} values will be converted to acute NOEC values by multiplying the effect concentration by 0.2.

- Interspecies adjustments will be made by multiplying the effect concentration by 0.1 (Newell et al. 1987). For species of different phylogenetic classes (e.g., mammal to bird), 0.05 will be used as the uncertainty factor.

When available, wildlife-specific dietary toxicity values will be compared to concentrations of specific constituents in the diet of the animal.

8.4 COMBINED HEALTH EFFECTS FROM MIXED WASTE

Sites that have both radioactive and chemical contaminants (mixed waste) present a unique set of potential risks: radiological carcinogenesis, nonstochastic effects of radiation, chemical carcinogenesis, and the noncarcinogenic effects of chemical toxicants. At present, governmental regulatory agencies have only marginally addressed the problem of quantifying the risks associated with mixed waste.

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8.4.1 Regulatory Guidance

To address this issue, current regulatory policies pertaining to health effects from mixed wastes, and toxicological assessments that may address these health effects, will be reviewed. In both cases, information is scarce or nonexistent, making definitive statements on methods for addressing this issue difficult.

In reviewing various regulations such as CERCLA, RCRA, and NESHAPS, it was found that no specific regulatory standards exist for estimating the combined risk from chemicals and ionizing radiation exposure in a mixed waste situation. However, the EPA has suggested that when cancer is the endpoint being evaluated, substance-specific cancer risks may be summed to determine a site-specific total risk (EPA 1989a). In addition, the EPA suggests that when both chemical and radiological standards have been set for a substance, the form with the strictest standard should be chosen. EPA risk assessment guidance also states that radiological and chemical risk estimates should be tabulated separately (EPA 1989a).

8.4.2 Health Effects from Exposures to Mixed Wastes

Review of the available literature addressing health effects from mixed wastes does not conclusively indicate additivity is the proper model to use to describe these effects. Little information is known about the interactions of ionizing radiation and chemicals. This interaction has best been documented in epidemiological studies of tobacco-users exposed to radiation (NAS 1988).

Studies of miners exposed to both tobacco smoke and radon have not yielded definitive results as to whether the interactions of these exposures are antagonistic, additive, or multiplicative (synergistic). Several small statistical studies have yielded mixed results. The largest study done by Whittemore and McMillan (1983) on Colorado uranium miners supported a multiplicative interaction. On the other hand, studies of Swedish miners exposed to radon daughters and followed for a long period of time did not show synergism between smoking and radon exposure (Radford and St. Clair Renard 1984). Studies on the A-bomb survivors provided no indication of interaction between smoking and ionizing radiation. In fact, both additive and multiplicative models fit the data obtained. However, these studies provide only limited data on addressing this interaction because the association of cancer with each of the factors individually is more complex than can be statistically documented.

The actual biological relationship between carcinogenesis and radiation exposure and/or smoking is characterized by interactions such as age at first exposure, sex, diet, and genetic predisposition. When studying the combined effects of cigarette smoking and radon exposure, factors such as the sequence of exposures and the degree exposures overlap becomes important. Unfortunately, most models do not account for these factors. The BEIR IV Committee reported that a sub-multiplicative model may be the best method of addressing these complicated interactions (NAS 1988).

The National Council on Radiation Protection (NCRP) reviewed the influence of environmental factors (in all cases, cigarette smoking) on radiogenic risk, and whether such factors interacted with ionizing radiation to increase or decrease cancer effects. (No studies on the combined effects from exposure to low-level radiation and chemicals were available for review.) In the four studies reviewed, the NCRP found that cigarette smoking affected radiation cancer in the following manner (NCRP 1989):

- Lung cancer data - inconclusive
- U.S. uranium miners (radon daughters) - synergistic effects
- Swedish iron miners - additive
- A-bomb survivors - additive

In perhaps the most extensive study addressing the issue of the differences between radiological and chemical risk, the NCRP (1989) stated that the <u>principles</u> for assessing carcinogenic risks of ionizing radiation and chemicals are <u>in essence</u> similar. However, differences exist. Issues involved in these differences are outlined below:

- Although the risks of ionizing radiation can be inferred from one radionuclide to another, chemicals vary widely in molecular structure, metabolism, mechanism of action, potency, and the stage in the cancer process during which they act. It has been argued that these differences make comparisons to radiation risk difficult. However, two responses to this argument exist. For both radionuclides and chemicals, carcinogenic effects have been noted in almost every organ of the body; no major differences in cancer distribution occur among both radionuclides and chemicals. In addition, although chemical carcinogens vary greatly in mechanism of action, metabolism, etc., they have historically been compared among each other.

- Historically, risk from exposure to ionizing radiation has been calculated for exposures above background levels. Although in the past risks calculated for chemical carcinogens have been absolute values, the move toward calculating the risk above background exposure has begun.

- Of the 3500 potential carcinogens identified by the National Academy of Sciences (NAS 1984), only 23 have been verified as human carcinogens by the International Agency for Research on Cancer (IARC 1982). Ionizing radiation has been shown to be a human carcinogen. This is perhaps the greatest difference in comparing chemical carcinogens and radionuclides.

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- Approximately 23 chemicals are known to cause cancer in man. (EPA only lists 10 Class A carcinogens.) In these cases, epidemiological data have been used to estimate human risks using a linear model, as is the case with radiation carcinogenesis. In both cases, the only extrapolation required is from high occupational doses to low environmental doses.

- Hundreds of chemicals have been identified as carcinogens in laboratory animals. To infer risk using these studies requires extrapolation between small rodents to humans using the linearized multistage model, and extrapolation from near toxic doses to low environmental doses. However, according to recent studies (Rowe and Springer 1986), the human health risks estimated using animal data closely match human risks estimated using data from epidemiology studies. Radiological risk evaluation does not depend exclusively on interspecies extrapolation. Radiological risk evaluation is primarily based on a large cohort of human A-bomb survivors.

8.4.3 Conclusions

Based on limited available information about combined effects from radiocarcinogenesis and chemical carcinogenesis, the following approach will be used for the FEMP risk assessments:

- Risk estimates for exposure to radionuclides will be tabulated separately from other contaminants.

- Risk estimates for radionuclides and chemical contaminants will be summed to determine the overall site risk whenever the same individuals are to be potentially exposed to both radionuclides and chemical contaminants.

- An explanation of uncertainties associated with combining risk estimates for radionuclides and chemical contaminants will be included in risk assessment reports.

8.5 UNCERTAINTIES RELATED TO TOXICITY INFORMATION

The uncertainties associated with the reference doses and slope factors used to quantify risk are well documented. Uncertainties include the use of uncertainty factors for noncarcinogens and the upper 95 percent confidence limit on the dose-response relationships for carcinogens, and the validity of using dose-response information from effects observed at high doses to predict adverse effects from exposure to low doses. These types of uncertainties will be documented qualitatively in the risk assessment reports.

Uncertainties related to ecological toxicity information are similar to those for human health toxicity information, with the additional factor that the receptors of concern belong to many species, rather than just one. The quality and design of studies are variable and can be difficult to compare. Laboratory studies of toxicity often use much higher doses of a chemical than those to which a receptor is likely to be exposed in the field. As in human health risk assessments, ecological risk assessments rely on extrapolation of results of studies between species that may vary in their sensitivity to a given chemical. Further uncertainty is introduced by the fact that receptors in the field are likely to be exposed to many constituents simultaneously, while toxicity data are usually based on exposures to one constituent. It is therefore difficult to assess the consequences of synergistic effects of exposure to mixtures of constituents. Finally, the controlled environment of the laboratory, necessary for reproducible experiments, eliminates many variables that may affect species' responses in the field. For example, organisms in the field may be able to reduce exposure to a toxicant by avoiding it, a response not available to them in the laboratory. Conversely, fathead minnows (Pimephales promelas) provided with territory (cover) required 1.83 mg/L zinc to elicit an avoidance response, but required only 0.284 mg/L when no territory was available (Korver and Sprague 1989). Comparable information is available for few toxicants.

9.0

9.0 RISK CHARACTERIZATION

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Risk characterization is the final step in the baseline risk assessment process, and involves combining the information developed in the toxicity assessment and the exposure assessment. This information is integrated and presented as qualitative and quantitative estimates of health risk. Risk characterization also supports the FS detailed analysis of alternatives, with short-term and long-term risks characterized for each alternative. Details concerning risk characterization for the FS risk assessments are presented in Section 10.

Potential carcinogenic effects are presented as the probability an individual will develop cancer over a lifetime of exposure, and are characterized by combining estimated intakes and dose-response information. The EPA has provided guidance for human health risk characterization, and the following documents will be used as major sources of guidance for preparing risk assessments for the FEMP: EPA 1991a, 1991c, 1991d, 1990a, 1990b, 1989a, 1989e, 1989g, 1988a, and 1984b.

9.1 RISK CHARACTERIZATION FOR RI BASELINE RISK ASSESSMENTS

Risks are characterized and evaluated quantitatively for current and future baseline conditions. As discussed in Sections 5.0, 6.0, and 7.0, information required from the exposure assessment includes:

- Exposure modeling assumptions
- Exposure pathway identification
- Estimated intakes

Information required from the toxicity assessment (Section 8.0) includes:

- Slope factors and weight of evidence classifications for all carcinogenic chemicals including the type of cancer
- Chronic and subchronic RfDs and shorter-term toxicity values and critical effects associated with each chemical
- Uncertainty and modifying factors and degree of confidence of RfDs
- Whether toxicity values are absorbed or administered doses
- Information that may affect animal-to-human or exposure route extrapolations
- NOECs for all chemicals for effects on ecological receptors

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9.2 RISK CHARACTERIZATION METHODOLOGY

Potential risks to humans following exposure to nonradioactive chemicals and radionuclides of potential concern are estimated using methods established by the EPA when available. Methods described by the EPA are health-protective and are likely to overestimate, rather than underestimate, risk. "For known or suspected carcinogens, acceptable exposure levels are generally concentration levels that represent an excess upper bound lifetime cancer risk to an individual of between 10^{-4} and 10^{-6} using information on the relationship between dose and response" (EPA 1990a).

9.2.1 Hazardous Chemical Exposures

Risks from hazardous chemicals are calculated for either carcinogenic or noncarcinogenic effects. Some carcinogenic chemicals also may pose a toxic (noncarcinogenic) hazard; risks from these chemicals will be characterized for both types of health effects.

9.2.1.1 Methodology for Carcinogens

The risk attributed to exposure to chemical carcinogens is estimated as the probability of an individual developing cancer over a lifetime as a result of exposure to a potential carcinogen. At low doses, the risk of developing cancer is determined as follows (EPA 1989a):

$$\text{Risk} = (\text{CDI})(\text{SF}) \qquad (9\text{-}1)$$

where

Risk = Risk of cancer incidence, expressed as a unitless probability
CDI = Chronic daily intake averaged over 70 years (mg/kg-day)
SF = Slope factor $(\text{mg/kg-day})^{-1}$

For a given pathway with simultaneous exposure of a receptor to several carcinogens, the following equation will be used to sum cancer risks:

$$\text{Risk}_P = \text{Risk}(\text{chem}_1) + \text{Risk}(\text{chem}_2) + \ldots \text{Risk}(\text{chem}_i) \qquad (9\text{-}2)$$

where

Risk_P = Total pathway risk of cancer incidence
chem_i = Individual carcinogenic chemical

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9.2.1.2 Methodology for Noncarcinogens

The risks associated with the effects of noncarcinogenic hazardous chemicals are evaluated by comparing an exposure level or intake to a reference dose. The ratio of intake over the reference dose is termed the Hazard Quotient (HQ) (EPA 1989a) and is defined as:

$$HQ = I/RfD \qquad (9\text{-}3)$$

where

HQ	=	Hazard quotient (unitless)
I	=	Intake of a chemical (mg/kg-day)
RfD	=	Reference dose (mg/kg-day)

When using this equation to estimate potential risk, both the intake and the RfD must refer to exposures of equivalent duration (i.e., sub-chronic, chronic, or less than two weeks). Chemical exposures are evaluated in all cases on a chronic basis, using chronic RfD values.

This approach is different from the probabilistic approach used to evaluate carcinogens. An HQ of 0.01 does not imply a 1 in 100 chance of an adverse effect, but indicates only that the estimated intake is 100 times less than the reference dose. An HQ of unity (1) indicates that the exposure intake is equal to the RfD. If the HQ is greater than 1 or "above unity", there may be concern for potential health effects.

In the case of simultaneous exposure of a receptor to several chemicals, a Hazard Index (HI) will be calculated as the sum of the Hazard Quotients by:

$$HI = I_1/RfD_1 + I_2/RfD_2 + \ldots I_i/RfD_i \qquad (9\text{-}4)$$

where

I_i	=	Intake for the i^{th} toxicant
RfD_i	=	Reference dose for the i^{th} toxicant

Hazard indices will be determined by assuming dose additivity for those chemicals acting by the same mechanism and inducing the same effects (EPA 1989a). Since we are assuming dose additivity, hazard quotients for chemicals that affect the same target organ will be summed.

9.2.2 Radiological Exposures

The radionuclide slope factors in HEAST, Table C, are the "maximum likelihood estimates of the age-averaged lifetime total excess cancer risk per unit intake or exposure" (EPA 1991a).

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Procedures for estimating the lifetime total excess cancer risks due to continuous, lifetime exposure (i.e., a 70-year average lifespan) to a radionuclide are discussed below.

In each case the slope factor simply acts as a "conversion factor" by which a radionuclide intake or a soil concentration is converted to the corresponding cancer risk in a single step. Cancer risks associated with the intake (inhalation and ingestion) of a radionuclide or with the concentration of a radionuclide in soil. Radiation doses to the whole body or to specific organs or tissues from such exposures cannot be readily calculated by use of slope factors.

9.2.2.1 Methodology for Internal Exposures

Risk characterization for internal exposures to radionuclides (intake via inhalation or ingestion) is calculated as follows:

$$\text{Risk} = (\text{I})(\text{SF}) \qquad (9\text{-}5)$$

where

Risk	=	Risk of cancer incidence, expressed as a unitless probability
I	=	Radionuclide intake (pCi)
SF	=	Radionuclide slope factor (pCi^{-1})

9.2.2.2 Methodology for External Gamma Exposures

Risk characterization for external exposure to gamma-emitting radionuclides in contaminated surface soil is calculated as follows:

$$\text{Risk} = (C_s)(\text{SF})(\rho)(\text{T})(\text{ED})(\text{MF})(\text{CF}) \qquad (9\text{-}6)$$

where

Risk	=	Risk of cancer incidence, expressed as a unitless probability
C_s	=	Radionuclide soil concentration (pCi/g)
SF	=	Radionuclide slope factor (risk/yr - pCi/m^2) [EPA 1991a]
ρ	=	Soil density (g/cm^3)
T	=	Soil depth (cm)
ED	=	Exposure duration (yr)
MF	=	Modifying factor, fraction of year exposed (unitless)
CF	=	Unit conversion factor = 1×10^4 cm^2/m^2

A soil density, ρ, of 1.5 g/cm^3 will be used as a site-specific value (USDA 1982). A soil depth, T, of 10 cm will be used for this calculation, in accordance with the methodology used in HEAST (EPA 1991a).

External slope factors do not include contributions from decay products (radioactive progeny). In some cases, these contributions can be substantial and will be factored into the risk calculations. For example, to estimate the total lifetime excess cancer risk due to continuous, lifetime external exposure to soil contaminated with radium-226 and its progeny (assuming secular equilibrium) will be calculated as the summation of the risks contributed by radium-226 and each decay product that emits photon radiation, such as lead-214 and bismuth-214.

Risk characterization for external exposures to gamma-emitting radionuclides in forms other than soil is calculated in the following manner:

$$\text{Risk} = (\text{DE})(\text{RC}) \qquad (9\text{-}7)$$

where

Risk	=	Risk of cancer incidence, expressed as a unitless probability
DE	=	Total dose equivalent (mrem) [from Equation 7-23]
RC	=	Cancer risk coefficient (mrem^{-1})

This methodology is used because the EPA slope factors method is not applicable to exposure scenarios involving gamma emissions from sources other than contaminated soils. For example, this methodology is useful for characterizing the risk from gamma-ray emissions from the K-65 silos. The cancer risk coefficient used is not radionuclide-specific; therefore, the same coefficient is used in all cases to which this method applies. As described in Section 8.2, the value of the risk coefficient is 6.2×10^{-7} mrem^{-1}.

9.3 PRESENTATION OF RISK CHARACTERIZATION RESULTS

The summary of risk characterization to be presented in each risk assessment report will include a tabulation of cancer risks and HIs associated with potential exposure pathways. The RME also will be assessed for all exposure pathways from the entire site under current and future land-use conditions. The calculated risks will also be presented in tabular form in the text. As described in Section 8.4, the risks of cancer induction by radionuclides and carcinogenic chemicals will be presented separately to reveal the magnitude of risk contributed by these two different types of contaminants at the site. The risks of cancer induction by radionuclides and carcinogenic chemicals will also be added to present the magnitude of cancer risk from all carcinogenic contaminants attributed to the site. An explanation of uncertainties due to adding risk estimates for radionuclides and chemical contaminants will be included in risk assessment reports.

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9.4 ECOLOGICAL RISK CALCULATIONS

This section describes how risks to ecological receptors at the FEMP will be characterized. The methodology used to estimate contaminant exposure and uptake is described above in Section 7.4.

9.4.1 Plants

Risk to vegetation as a result of exposure to radioactive and nonradioactive constituents in FEMP soils will be evaluated by comparison to plant toxicity data published in the literature. Maximum radiation doses and concentrations of nonradioactive constituents predicted in FEMP vegetation will be compared to the LOEC reported in the literature, with specific emphasis placed on adverse effects on reproduction and plant growth. When radiation doses or constituent concentrations in FEMP vegetation are predicted to exceed toxic levels reported in the literature, it will be concluded that constituent concentrations in FEMP soils may be hazardous to vegetation.

9.4.2 Terrestrial Animals

Risks of exposure of terrestrial animals to radiation will be assessed by comparing estimated doses to animals at the FEMP to values reported in the literature to cause chronic or acute effects. Risks from nonradiological constituents to terrestrial animals will be assessed based on literature toxicity data and the quotient method as described below. Concentrations of metals and inorganic substances predicted in animal muscle will be compared with concentrations in animals from other contaminated and noncontaminated sites, as reported in the literature, to indicate the relative extent of predicted contamination in FEMP wildlife.

To evaluate risks of chemical intake to each indicator species, intake values for a given constituent will be summed across pathways and compared to the NOEC and LOEC. As with the hazard quotient in human health risk assessments, if the quotient of the intake divided by the NOEC exceeds unity, it is concluded that the indicator species may be exposed to hazardous concentrations of a given constituent at the FEMP. Quotients will be summed for chemicals with similar modes of action and a "hazard index" calculated. If either the quotient or hazard index is less than one, the species is not expected to be exposed to any adverse effects via the soil and vegetation ingestion pathways.

9.4.3 Aquatic Organisms

Risks from exposure of aquatic organisms to radiation will be assessed by comparing estimated doses to organisms in surface waters at and adjacent to the FEMP to values reported in the literature to cause chronic or acute effects (e.g., EPA 1986b, 1988d, 1988e). Risks to aquatic organisms from nonradiological constituents will be assessed based on literature toxicity data for

NOECs and LOECs and EPA and OEPA acute and chronic water quality standards (EPA 1986a, OEPA 1990b). If the ratio of the predicted average concentration of a constituent to the NOEC or water quality standard exceeds one, it will be concluded that aquatic organisms in the water body of concern may be exposed to toxic levels of the constituent. OEPA standards will be used for all constituents for which they exist. If a EPA standard exists for any of the remaining constituents, it will be used. Literature values for the NOEC will be used only for those constituents lacking an OEPA or EPA standard.

Characterization of present risks from FEMP constituents to aquatic organisms will also incorporate the results of RI/FS studies focussed on them. Field and laboratory work supporting these studies has been completed and the results are currently undergoing internal technical review. The benthic macroinvertebrate communities of Paddys Run and the Great Miami River have been surveyed five times over two years, 1988 to 1990, comparing sampling sites upstream, adjacent to, and downstream from FEMP influence. Data analyses include species abundances, diversity and evenness, tolerance indices (Weber 1973), and OEPA's Invertebrate Community Index (OEPA 1988).

The effects of the existing NPDES-permitted discharge from the FEMP to the Great Miami River have been examined using standard EPA acute and chronic toxicity tests (Peltier and Weber 1985, Weber et al. 1989). The results of these tests will be compared with the effluent composition at the time of sampling, as reported to OEPA and DOE by WEMCO, to estimate the potential effects of FEMP effluent on aquatic organisms in the Great Miami River.

Finally, the aquatic toxicity of water-extractable substances from soils and sediments at the FEMP has been examined using acute toxicity tests. These tests provide an indication of the potential effects of leachate and runoff from FEMP soils and sediments on aquatic organisms.

9.5 UNCERTAINTIES ASSOCIATED WITH RISK ASSESSMENTS

Uncertainties in risk assessments for the FEMP will be presented as a conditional estimate independently based on a number of assumptions regarding exposure and toxicity. The assumptions and uncertainties will be fully specified in each risk assessment and both qualitative and quantitative evaluation of uncertainties will be performed.

It is not anticipated that a highly quantitative statistical analysis of uncertainties can be performed due to the nature and scope of risk assessments under CERCLA. As with all other environmental risk assessments, the uncertainty about the numerical results of the risk assessments at the FEMP is anticipated to be a factor of ten or greater. The individual contributions to this uncertainty will be discussed in each risk assessment report.

Site-related assumptions and parameters will be evaluated to determine which of these contribute significantly to the overall uncertainty of the assessment. The assumptions and parameters that contribute most significantly to the uncertainty will be investigated to determine which can be defined more precisely to reduce the uncertainty.

Major sources of uncertainty can be grouped into four categories. These are: definition of physical setting; applicability and assumptions for models; parameter values for fate, transport and exposure; and toxicity and risk characterization.

Within the definition of the physical setting, uncertainties will be presented for inclusion/exclusion of chemicals having a quantitative risk assessment, assumptions and parameters for current and future land use, and inclusion/exclusion of exposure pathways. Uncertainties associated with the selection of multiple exposure pathways for the RME scenario will be discussed.

An evaluation of the appropriateness of the exposure models and their mathematical formulation for the FEMP will be presented as part of the uncertainty analysis. The key assumptions used in the models will be listed and explained, along with a discussion of the potential impact of each on the risk calculation.

Fate, transport, and exposure parameter values will be listed, including numerous values presented in Sections 6.0 and 7.0. If possible, the uncertainty analysis of each risk assessment will describe measured or assumed parameter value distributions. The potential magnitude and direction of bias (i.e., overestimation or underestimation of risk) resulting from assumptions and parameter values will be described in tabular form in the risk assessment.

Uncertainties in toxicity and risk characterization will be evaluated with respect to the assumptions for derivation of toxicity values, potential for interactions from multiple chemicals. An evaluation of the uncertainty due to exclusion of chemicals or radionuclides from the quantitative risk assessment will be presented.

Perhaps the greatest uncertainties are associated with calculation of risks from multiple contaminants in multiple source areas with multiple exposure pathways from the FEMP. As stated previously, carcinogenic risks from multiple contaminants will be presented separately (by contaminant and pathway) and will be combined (added) for hypothetical receptors at each specified location. Similarly, noncancer hazard indices will be presented separately (by contaminant and pathway) and will be combined (added) for hypothetical receptors at each specified location. The uncertainty in calculated risks as a consequence of these assumptions will be discussed.

A semi-quantitative analysis of uncertainties will be performed for risk assessments at the FEMP. The potential range of values associated with each assumption or parameter will be presented. A sensitivity analysis will be performed to estimate the range of risks that result from combinations of assumptions and parameters.

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10.0 RISK ASSESSMENT/RISK MANAGEMENT IN THE FEASIBILITY STUDY PROCESS

Risk assessment/risk management support in the feasibility study process can be divided into three major tasks:

- Development of Remedial Action Objectives (RAOs) and Preliminary Remediation Goals (PRGs)
- Evaluation of the risks associated with remedial alternatives for each operable unit
- Management and optimization of risks from a site-wide perspective

Each of these tasks will be described in this section.

10.1 REMEDIAL ACTION OBJECTIVES

After completion of the RI and prior to the beginning of the evaluation of alternatives, RAOs and PRGs must be established. These goals will be used by engineers as design criteria during the alternative development and selection process. RAOs are site-specific, qualitative goals that define the extent of cleanup required to achieve a CERCLA response action (EPA 1988a). RAOs address contaminants of concern, media of concern, potential exposure pathways and remediation goals (EPA 1990a).

No precedent exists for developing RAOs and PRGs for a mixed waste CERCLA site, perhaps with the exception of work performed at the Maxey Flats Disposal Site (see EPA 1991e). In addition, specific guidance for developing RAOs is not yet available from the EPA. A review of the draft document, Risk Assessment Guidance for Superfund: Volume 1 - Human Health Evaluation Manual, Part B, Development of Preliminary Remediation Goals, (RAGS, Part B), which gives guidance on refinement of remediation goals indicates that the document does not address mixed waste issues.

10.1.1 Preliminary Remediation Goals

PRGs are chemical-specific, medium-specific numerical concentration limits that should address all contaminants and all pathways found to be of concern during the baseline risk assessment process. Remediation goals are defined in the NCP at 40CFR300.430(e)(2)(i) as:

"(A) Applicable or relevant and appropriate requirements under federal environmental or state environmental or facility siting laws, if available, and the following factors:

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1) For systemic toxicants, acceptable exposure levels shall represent concentration levels to which the human population, including sensitive subgroups, may be exposed without adverse effect during a lifetime or part of a lifetime, incorporating an adequate margin of safety

2) For known or suspected carcinogens, acceptable exposure levels are generally concentration levels representing an excess upper bound lifetime cancer risk to an individual of between 10^{-4} and 10^{-6} using information on the relationship between dose and response. The 10^{-6} risk level shall be used as the point of departure for determining remediation goals for alternatives when ARARs are not available or are not sufficiently protective because of the presence of multiple contaminants at a site or multiple pathways of exposure

3) Factors related to technical limitations such as detection/quantification limits for contaminants

4) Factors related to uncertainty

5) Other pertinent information

(B) Maximum contaminant level goals (MCLGs) established under the Safe Drinking Water Act, that are set at levels above zero, shall be attained by remedial actions for ground or surface waters that are current or potential sources of drinking water, where the MCLGs are relevant and appropriate under the circumstances of the release based on the factors in § 300.400(g)(2)[2]. If an MCLG is determined not to be relevant and appropriate, the corresponding maximum contaminant level (MCL) shall be attained where relevant and appropriate to the circumstances of the release.

(C) Where the MCLG for a contaminant has been set at a level of zero, the MCL promulgated for that contaminant under the Safe Drinking Water Act shall be attained by remedial actions for ground or surface waters that are current or potential sources of drinking water, where the MCL is relevant and appropriate under the circumstances of the release based on the factors in § 300.400(g)(2).

(D) In cases involving multiple contaminants or pathways where attainment of chemical-specific ARARs will result in cumulative risk in excess of 10^{-4}, criteria in paragraph (e)(2)(i)(A) of this section may also be considered when determining the cleanup level to be attained.

(E) Water quality criteria established under sections 303 or 304 of the Clean Water Act shall be attained where relevant and appropriate under the circumstances of the release.

(F) An alternate concentration limit (ACL) may be established in accordance with CERCLA section 121(d)(2)(B)(ii).

(G) Environmental evaluations shall be performed to assess threats to the environment, especially sensitive habitats and critical habitats of species protected under the Endangered Species Act" (EPA 1990a).

Guidance is available from the EPA for developing risk-based PRGs (EPA 1991f). PRGs are developed early in the RI/FS process. They are dependent on the identification of ARARs as well as on the knowledge of the risk assessment process (EPA 1991f).

Guidance published in the preamble of the NCP states that PRGs should be based on readily available environmental or health-based ARARs, ambient water quality criteria, and other criteria, advisories or guidance (EPA 1990a). Many identified ARARs have not been derived from risk levels that would meet the CERCLA objectives of "protectiveness of human health". In other words, PRGs based on ARARs could be less stringent than criteria based on the 10^{-4} to 10^{-6} risk level. However, ARARs are considered to be acceptable as action levels in the CERCLA process (EPA 1991f).

ARARs do not exist for many chemicals in various environmental media. For these chemicals, risk-based PRGs will be developed. Risk-based PRGs will be used as initial guidelines. They do not establish final cleanup goals (EPA 1991f).

At the FEMP, a single set of initial PRGs will be developed and used for each operable unit in the early stages of screening alternatives. Because the initial PRGs will be generic for the site, and not operable unit-specific, they will be based on generic default exposure pathways and equation assumptions recommended by the EPA in Risk Assessment Guidance for Superfund: Volume 1 - Human Health Evaluation Manual, Part B, Development of Preliminary Remediation Goals and the exposure parameters presented in this document. These pathways are considered to be "limiting" pathways, viz., pathways that often are responsible for much of the baseline risk.

These PRGs will be formally presented in the Site-Wide Characterization Report. However, as suggested by EPA (1991f) a memorandum containing initial PRGs will be distributed to the RPM, project managers and project engineers as soon as possible.

Initial PRGs may need to be modified as operable unit-specific baseline risk assessments are completed. Thus, in using initial PRGs in the early stages of the alternative screening process, engineers should understand that PRGs may be modified and should make the design of alternatives flexible. Chemicals may be added or deleted from the list of chemicals of concern, or

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PRGs may need to be modified based on the identification of additional limiting pathways. Modified PRGs will be presented in the operable unit feasibility study reports.

PRGs are refined into final remediation levels and presented in the Record of Decision. Final remediation levels must meet "the threshold criteria" of "protection of human health and the environment" and "compliance with ARARs", but may be modified "based on the balancing and modifying criteria and factors relating to uncertainty, exposure and technical feasibility," (EPA 1990a).

Note should be taken that, with the exception of recommending the inclusion of environmental ARARs in the selection of PRGs, RAGS, Part B addresses human health effects. Available environmental ARARs, e.g., Water Quality Criteria from the Clean Water Act, will be incorporated into the selection process; however, specific environmental risk concerns will be addressed as PRGs are modified based on the results of the operable unit-specific ecological risk assessments.

10.1.2 Methodology for Risk-Based PRGs

Development of initial risk-based PRGs requires the following information:

- Chemicals of potential concern
- Environmental media of potential concern
- Probable future land use
- Chemical-specific toxicity information
- Target risk levels

Information on the chemicals of potential concern and environmental media of potential concern will be determined as stated in Section 4.0 and Section 5.0, respectively. Probable future land use is described in Section 5.0. To develop PRGs, it is assumed that the future land use scenario is the resident farmer. Toxicity data used to develop PRGs are discussed in Section 9.0. In general, cancer slope factors and reference doses from IRIS and HEAST will be used.

10.1.2.1 Target Risk

In developing risk-based PRGs, target risk levels (TR) must be established for carcinogens and a target hazard quotient (THQ) and target hazard index (THI) (the sum of the THQs) must be established for noncarcinogens. Once these levels are established, they can be used in conjunction with toxicity data and exposure equations to calculate PRGs.

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One of the goals of the NCP is to manage total site-wide risks such that the sum of all risks does not exceed 10^{-4}. EPA suggests a default target risk of 10^{-6} (EPA 1991f). This risk, 10^{-6}, will be used as a target risk for the FEMP PRGs. In addition, PRGs will also be developed using a target risk of 10^{-5}. The availability of the range of PRGs provides useful information for eventual cost-benefit analysis as part of the remedy selection process.

The EPA indicates that the cumulative site HI should be less than 1. However, while total noncancer risk cannot exceed an HI of 1, no direct guidance is available on apportioning the allowable level among the various chemicals in the various environmental media. The most applicable regulatory guidance comes from the Office of Drinking Water (ODW), which, in calculating Maximum Contaminant Level Goals (MCLG), uses a relative source contribution (RSC) factor to account for the contribution from other sources of exposure (EPA 1989h). If sufficient data are not available to evaluate the drinking water exposure relative to other exposures, ODW assumes other exposures account for 80 percent of the total, leaving 20 percent for water. Thus the default RSC is 20 percent (0.20).

This method can be adapted to the development of PRGs for noncarcinogens. Because it is not known what additional sources are contributing to total exposure, the default RSC of 0.20 will be used to develop individual chemical/media specific PRGs, helping to insure that the total HI from each exposure does not exceed 1. Thus, the THQ for medium-specific, noncarcinogenic effects will be 0.2, helping to insure the THI is less than or equal to 1, as recommended by EPA (1991f).

10.1.2.2 <u>Groundwater Exposures</u>

Because the NCP encourages protection of groundwater for its maximum use, and because the future land-use scenario at the FEMP assumes a resident farmer may use groundwater in the deep aquifer as potable water, risk-based PRGs will be calculated assuming groundwater as potable water. EPA suggests using potable water use, drinking water, and gaseous emission while showering as default exposure pathways for determining PRGs (EPA 1991f). Although additional pathways may exist, these represent the most reasonable and potentially limiting pathways. Equations 10-1 through 10-4 address these pathways. At the FEMP, volatile compounds are not present in the aquifer or in the waste unit sources in sufficient quantities to warrant evaluating volatilization from showering. Thus, the drinking water pathway will be the sole exposure pathway to develop PRGs for organic compounds, inorganics and radionuclides (except radon). Volatilization will be used to develop radon PRGs.

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Most of the parameters presented in the following exposure equations are available in Sections 6.0 and 7.0. Parameters not defined in this section will be defined as they are presented.

For noncarcinogens, the exposure equation is:

$$C_w = (THI)(BW)(AT)(365\ \text{days/yr})/(EF)(ED)[(1/RfD_o)(IR_w)] \qquad (10\text{-}1)$$

where

C_w	=	PRG concentration in water (mg/L)
THI	=	Target Hazard Index (1)
RfD_o	=	Oral reference dose (mg/kg/day)
BW	=	Adult body weight (kg)
AT	=	Averaging time (yr)
EF	=	Exposure frequency (days/yr)
ED	=	Exposure duration (yr)
IR_w	=	Daily water ingestion rate (L/day)

For chemical carcinogens, the exposure equation is:

$$C_w = (TR)(BW)(AT)(365\ \text{days/year})/(ED)(EF)[(SF_o)(IRw)] \qquad (10\text{-}2)$$

where

C_w	=	PRG concentration in water (mg/L)
TR	=	Target risk (1×10^{-5} and 1×10^{-6}).
BW	=	Adult body weight (kg)
AT	=	Averaging time (yr)
EF	=	Exposure frequency (days/yr)
ED	=	Exposure duration (yr)
SF_o	=	Oral slope factor $(mg/kg/day)^{-1}$
IR_w	=	Daily water ingestion rate (L/day)

For radionuclides, with the exception of radon the exposure equation is:

$$C_w = (TR)/(EF)(ED)(SF_o)(IR_w) \qquad (10\text{-}3)$$

where

C_w	=	PRG concentration in water (pCi/L)
TR	=	Target risk (1×10^{-5} and 1×10^{-6}).
EF	=	Exposure frequency (days/yr)
ED	=	Exposure duration (yr)

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SF_o = Oral slope factor (risk/pCi)
IR_w = Daily water ingestion rate (L/day)

For radon the exposure equation is:

$$C_w = (TR)/(EF)(ED)[(SF_o)(IR_w)+(SF_i)(K)(IR_a)] \qquad (10\text{-}4)$$

where

C_w = PRG radionuclide concentration in water (pCi/L)
TR = Target risk (1×10^{-5} and 1×10^{-6}).
EF = Exposure frequency (days/yr)
ED = Exposure duration (yr)
SF_o = Oral slope factor (risk/pCi)
IR_w = Daily water ingestion rate (L/day)
SF_i = Inhalation slope factor (risk/pCi)
IR_a = Daily indoor inhalation rate (m^3/day)
K = Volatilization factor (L/m^3)

The volatilization factor (K) is a default value of 0.0005 x 1000 L/m^3 (Andelman 1990).

EPA recently published a proposed rule for radionuclides in drinking water (EPA 1991g). In it, EPA presented its findings on estimated cancer risks from radon in domestic water. It was estimated that 1.5 pCi/L corresponds to a 10^{-6} lifetime risk from radon via all water pathways. This published risk number will be compared with the value generated by the application of Equation 10-4, and the more conservative value will be selected as the PRG for radon in water.

10.1.2.3 Exposures to Perched Water

PRGs for perched water that is deemed usable for potable water will be based on equations 10-1 through 10-4 for groundwater exposures. However, many of the perched zones at the site are of limited area extent and have low hydraulic conductivity, leading to low yield rates. These zones can not be relied upon as year-round potable water sources. In general, typical rates for potable water wells are 200 gallons/day sustained yield (California State Water Resources Control Board) to 400 gallons/day for a family of four (Henderson and Jones 1982 and Reid 1965).

For perched water that is not a potential potable water source PRGs will be developed based on the potential for chemicals in those perched zones to leach into the bedrock aquifer or a receiving surface water body, thus equating water in the shallow zones to "leachate". Leachate is regulated

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by the U.S. EPA under 40 CFR 261 with the use of the Toxic Characteristic Leaching Potential (TCLP). TCLP regulatory levels are based on the acceptable drinking water concentrations multiplied by a dilution attenuation factor (DAF) which accounts for the degree of attenuation and dilution that a compound is expected to undergo during transport to the drinking water aquifer or receiving stream (EPA 1986c).

Both risk-based and ARAR-based acceptable drinking water concentrations will be used to develop PRGs for the perched waters. These values will be multiplied by the default DAF of 100 (EPA 1986c).

10.1.2.4 Soils and Waste Materials

PRGs for soils and waste materials will be developed using two distinct methods. The first method assumes that direct contact will occur with the contaminated material. The second assumes that the material is a source for future potential contamination in the groundwater. EPA has developed several models for use in determining soil clean-up levels based on potential contaminant migration to the groundwater and acceptable groundwater concentrations (EPA 1989i).

Application of each method greatly depends on the quantity of material in the soils or waste unit. If small quantities are being addressed (e.g., residual soil contamination in Operable Unit 5), the soil ingestion model is most applicable. For Operable Unit 1 pits, the Summers model (Summers et al. 1980) will likely be used.

EPA suggests that for residential land use, PRGs should be based on direct ingestion (EPA 1990c and EPA 1991f). In addition, since it is assumed that a resident farmer may plow the land annually, there is potential for disturbed soils to result in volatile and particulate emissions to the air. For radionuclides, direct external radiation exposures will also be considered. Equations 10-5 through 10-12 present the calculated methodology for determining PRGs for soils.

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For volatile organic noncarcinogenic effects, the exposure equation is:

$$C_s = \frac{(THI)\,(BW)\,(AT)\,(365 days/yr)}{(EF)\,(ED)\,[((1/RfD_o)\,(10^{-6} kg/mg)\,(IR_s)) + ((1/RfD_i)\,(IR_a)\,(1/VF + 1/PEF))]} \quad (10\text{-}5)$$

where

C_s	=	PRG concentration in soil (mg/kg)
THI	=	Target Hazard Index (0.20)
RfD_o	=	Oral reference dose (mg/kg/day)
RfD_i	=	Inhalation reference dose (mg/kg/day)
BW	=	Adult body weight (kg)
AT	=	Averaging time (yr)
EF	=	Exposure frequency (days/yr)
ED	=	Exposure duration (yr)
IR_s	=	Daily soil ingestion rate (mg/day)
IR_a	=	Daily inhalation rate (m^3/day)
VF	=	Soil-to-air volatilization factor (m^3/kg)
PEF	=	Particulate emission factor (m^3/kg)

Methods for evaluating volatilization and particulate emission factors are available from EPA (1991f). The method requires data input that may not be readily available, e.g., molecular diffusivity. If input data for volatilization and particulate emission calculations are not readily available, PRGs will be based on the ingestion pathway. For nonvolatile organics and inorganic noncarcinogenic effects, Equation 10-5 may be used without the expression for volatilization (1/VF).

For volatile organic chemical carcinogens, the exposure equation is:

$$C_s = \frac{(TR)\,(BW)\,(AT)\,(365 days/yr)}{(EF)\,(ED)\,[((SF_o)\,(10^{-6} kg/mg)\,(IR_s)) + ((SF_i)\,(IR_a)\,(1/VF + 1/PEF))]} \quad (10\text{-}6)$$

where

C_s	=	PRG concentration in soil (mg/kg)
TR	=	Target risk (1×10^{-5} and 1×10^{-6}).
BW	=	Adult body weight (kg)
AT	=	Averaging time (yr)
EF	=	Exposure frequency (days/yr)
ED	=	Exposure duration (yr)
SF_o	=	Oral slope factor $(mg/kg/day)^{-1}$

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SF_i	=	Inhalation slope factor $(mg/kg/day)^{-1}$
IR_s	=	Daily soil ingestion rate (mg/day)
IR_a	=	Daily inhalation rate (m^3/day)
VF	=	Soil-to-air volatilization factor (m^3/kg)
PEF	=	Particulate emission factor (m^3/kg)

For nonvolatile organics and inorganic carcinogens, Equation 10-6 may be used without the expression for volatilization (1/VF).

For radionuclides, the exposure equation is:

$$C_w = \frac{(TR)}{(ED)\,[((SF_o)(10^{-3})(EF)(IR_s)) + ([(SF_i)(10^{3})(EF)(IR_a)]\,[(1/VF) + (1/PEF)] + [(SF_e)(10^{3})(D)(SD)(1-S_e)(T_e)])} \quad (10\text{-}7)$$

where

C_w	=	PRG concentration in soil (pCi/g)
TR	=	Target risk (1×10^{-5} and 1×10^{-6}).
EF	=	Exposure frequency (days/yr)
ED	=	Exposure duration (yr)
SF_o	=	Oral slope factor (risk/pCi)
SF_i	=	Inhalation slope factor (risk/pCi)
SF_e	=	External exposure slope factor (risk/yr per pCi/m^2)
IR_s	=	Daily soil ingestion rate (mg/day)
IR_a	=	Daily inhalation rate (m^3/day)
VF	=	Soil-to-air volatilization factor (m^3/kg)
PEF	=	Particulate emission factor (m^3/kg)
D	=	Depth of radionuclide in soil (m)
SD	=	Soil density (kg/m^3)
S_e	=	Gamma shielding factor (unitless)
T_e	=	Gamma exposure time factor (unitless)

Depth in soil (D) is assumed to equal 0.1 meter (EPA 1991f). Time use modifying factors (MF) from Section 7.0 will be used to define T_e.

For nonvolatile radionuclides, Equation 10-7 may be used without the expression for volatilization (1/VF).

In addition, the revised Summers Model (Summers et al. 1980) will be used to calculate PRGs given the potential for soil to leach to the groundwater. The Summers Model is described below. The concentration of a chemical in groundwater is a function of the amount of the chemical

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infiltrating through the soil column to the aquifer and the amount of the chemical already present in the aquifer. The chemical concentration is also determined by the volume of water into which the leachate is dissolved. The equation for the Summers Model is:

$$C_w = \frac{(Q_p C_p) + (Q_a C_a)}{Q_p + Q_p} \qquad (10\text{-}8)$$

where

C_w	=	PRG concentration in water (mg/l)
Q_p	=	Volumetric flow rate of infiltration into the aquifer (ft^3/day), where $Q_p = V_{dz} \bullet A_p$, and (10-9)
V_{dz}	=	Darcy velocity in downward direction (ft/day)
A_p	=	Horizontal area of spill (ft^2)
C_p	=	Concentrations of pollutant in the infiltration at the unsaturated-saturated zone interface (µg/l)
Q_a	=	Volumetric flow rate of groundwater (ft^3/day), where $Q_a = Vd \bullet h \bullet w$, and (10-10)
V_a	=	Darcy velocity in aquifer (ft/day)
h	=	Aquifer thickness (ft)
w	=	Width of spill perpendicular to flow direction in aquifer (ft)
C_a	=	Initial or backward concentration of pollutant in aquifer (mg/l)

V_{dz} is estimated as the average annual precipitation minus surface runoff and evapotranspiration for the area, assuming all precipitation infiltrated through the soil.

The Darcy velocity in the aquifer (V_d) is estimated by:

$$V_d = KI \qquad (10\text{-}11)$$

where

K	=	Hydraulic conductivity (ft/day)
I	=	Hydraulic gradient (unitless)

It was assumed that the background concentrations of the chemicals (C_a) were equal to zero, and equations were rearranged to solve for C_s, the PRG concentration in soil:

$$C_s = \frac{C_{gw}(Q_p + Q_a)K_d}{Q_p} \qquad (10\text{-}15)$$

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where

K_d = Soil/water equilibrium partitioning coefficient (mL/g)

The above equations will be incorporated into a spreadsheet model to perform the calculations for all chemicals of interest.

The revised Summers Model makes the following assumptions:

- The soil/water system is at equilibrium
- No contaminant degradation is occurring
- The unsaturated soil zone is homogeneous down to the aquifer and
- Contaminants are mixed throughout the depth of the aquifer beneath the contaminant source

The model does not account for any contaminant dilution or attenuation due to horizontal transport within the aquifer. Acceptable soil concentrations are therefore determined based on the assumption that groundwater must meet acceptable or target levels within the aquifer directly beneath the source.

10.1.2.5 Presentation of PRGs

Presentation formats for PRGs suggested by EPA (1991f) will be modified to provide more detail and additional information. Tables 10-1 through 10-3 are example PRG presentation tables that, when completed for all chemicals of concern, will be sent to the RPM and site project managers and engineers, and will be included in the Site-Wide Characterization Report.

In addition to providing risk-based PRGs and ARARs, the tables provide background concentrations and Contract Laboratory Required Detection Limits. These concentrations act as reference points for understanding verification limitations of PRGs.

Two types of ARARs exist for radionuclides: chemical-specific radionuclide concentration limits (e.g., 5 pCi/L radium in drinking water [40CFR141]) and radiation dose limits (e.g., 100 mrem/yr [10CFR20]). Both types of ARARs must be considered. Existing chemical-specific concentration limits would be used for a radionuclide in a specified medium. Once all chemical-specific ARARs are accounted for and subtracted from the allowable dose limit, the remaining dose limit, if any, would be apportioned to radionuclides in media that have not been addressed by a chemical-specific ARAR.

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TABLE 10-1
EXAMPLE TABLE FORMAT FOR PRELIMINARY REMEDIATION GOALS - DEEP AQUIFER

Site: Fernald Environmental Management Project
Location: Fernald, Ohio
Medium: Groundwater

Land Use: Resident Farmer
Exposure Route: Water Ingestion

	RISK-BASED PRGs			ARAR/TBC-BASED PRGs FEDERAL STANDARDS		ARAR/TBC-BASED PRGs STATE STANDARDS	OTHER CONSIDERATIONS		
CHEMICAL	HI=1 RfD-Based PRG[a] (mg/L)	10^{-5} CSF-Based PRG[a] (mg/L)	10^{-6} CSF-Based PRG[a] (mg/L)	MCL[b] (mg/L)	MCLG[c] (mg/L)	Ground-Waters[d] (mg/L)	Back-ground[e] (mg/L)	Contract Required Detection Limit[f] (mg/L)	Range of Detected Concentrations in Deep Aquifer at the Site[g] (mg/L)
RADIONUCLIDES									
INORGANICS									
ORGANICS									

[a] See Section 10.1.2.
[b] Safe Drinking Water Act, 40 CFR 141, 142, Maximum Contaminant Levels for Drinking Water. Values denoted with and asterisk (*) indicate preliminary MCLs.
[c] Proposed Maximum Contaminant Level Goals, from individual Federal Registers as noted.
[d] Based on OAC 3745-81-16; For many radionulcides, values are based on an average annual dose of beta particle and photon (e.g., gamma) of 4 mrem/year.
[e] Upper 95%-tolerance interval of background concentrations from Shandon Trough RI/FS monitoring data (as suggested by EPA 1989c).
[f] From CLP Statement of Work 3900LM01.08.
[g] From "Nature and Extent of Contamination," Site-Wide Characterization Report, for 2000, 3000, and 4000 series wells.
• Indicates current maximum detected concentration in 2000-4000 series wells is above selected PRG.

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TABLE 10-2
EXAMPLE TABLE FORMAT FOR PRELIMINARY REMEDIATION GOALS - SURFACE SOILS AND WASTE MATERIAL

Site: Fernald Environmental Management Project
Location: Fernald, Ohio
Medium: Waste Materials

Land Use: Resident Farmer
Exposure and Transport Routes: Leaching to Groundwater, Direct Soil Ingestion

CHEMICAL	PRG based on Summers Leaching Model; assuming GW PRG[a] (mg/kg)	RISK-BASED PRGs: HI=1 RfD-Based PRG[b] (mg/kg)	RISK-BASED PRGs: 10^{-5} CSF-Based PRG[b] (mg/kg)	RISK-BASED PRGs: 10^{-6} CSF-Based PRG[b] (mg/kg)	ARAR/TBC PRGs[c] (mg/kg)	OTHER CONSIDERATIONS: Background[d] (mg/kg)	OTHER CONSIDERATIONS: Contract Required Detection Limit[e] (mg/kg)	Range of Concentration Detected in Soil/Waste Material[f] (mg/kg)
RADIONUCLIDES								
INORGANICS								
ORGANICS								

[a] U.S. EPA has developed several models for use in determining soil clean-up levels based on potential contaminant migration to the groundwater and acceptable groundwater concentrations (EPA 1989i). The revised Summer's model is used to determine risk-based clean-up levels in soil, based on acceptable groundwater concentrations (Summers 1980).

[b] See Section 10.1.2.

[c] Sources as noted for individual constituents.

[d] Upper 95%-tolerance interval from USGS (Shacklett 1984) and Myrick et al. (1983).

[e] From CLP Statement of Work, U.S. EPA OLM01.08.

[f] From RI/FS sampling.

• Indicates current maximum detected concentration in soil is above selected PRG.

TABLE 10-3
EXAMPLE TABLE FORMAT FOR PRELIMINARY REMEDIATION GOALS - PERCHED WATER ZONES

Site: Fernald Environmental Management Project
Location: Fernald, Ohio
Medium: Perched Water

Land Use: Resident Farmer
Transport Routes: Leaching to Groundwater

	RISK-BASED PRGs BASED ON REGULATING AS LEACHATE[a]			ARAR-BASED PRGs BASED ON REGULATING AS LEACHATE[a]		OTHER CONSIDERATIONS		
CHEMICAL	HI=1 RfD-Based PRG[b] (mg/L)	10^{-5} CSF-Based PRG[c] (mg/L)	10^{-6} CSF-Based PRG[c] (mg/L)	MCL-Based Limit[d] (mg/L)	TCLP Regulatory Limit[e] (mg/L)	Background[f] (mg/L)	Contract Required Detection Limit[g] (mg/L)	Range of Concentrations Detected in Perched Water[h] (mg/L)
RADIONUCLIDES								
INORGANICS								
ORGANICS								

[a] In the shallow water-bearing zones, PRGs are developed based on the potential for chemicals in those zones to leach into the bedrock aquifer or a receiving surface water body, thus equating water in the shallow zones to "leachate". Leachate is regulated by the U.S. EPA under 40 CFR 261 with the use of the Toxic Characteristic Leaching Potential (TCLP). TCLP regulatory levels are based on the acceptable drinking water concentrations multiplied by a dilution attenuation factor (DAF), which accounts for the degree of attenuation and dilution that a compound is expected to undergo during transport to the drinking water aquifer or receiving stream (EPA 1986c; 51FR21650).
[b] Based on HI-based groundwater PRG and a DAF of 100.
[c] Based on cancer risk-based groundwater PRG and a DAF of 100.
[d] Based on MCL times DAF of 100.
[e] From 40 CFR 261, 55FR11798.
[f] Upper 95%-tolerance interval from USGS (Shacklett 1984) and Myrick et al. (1983).
[g] From CLP Statement of Work, U.S. EPA OLM01.08.
[h] From RI/FS sampling.
• Indicates current maximum detected concentration in 1000 series wells is above selected PRG.

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10.1.3 Final Remediation Levels

While PRGs are developed early in the RI/FS process (prior to complete site characterization), final RGs are developed after an alternative has been selected (EPA 1990a). Final RGs are, in effect, cleanup levels that must be achieved by the selected technology. While PRGs will be based on preliminary risk information and default exposure equations, other factors may be considered in the development of the final goals. A major consideration will be identified ARARs. Other considerations that will play a role in selecting final RGs include:

- Technological feasibility
- Verification
- Uncertainties in risk estimates
- Historical precedence
- Acceptable risk

10.1.3.1 Technological Feasibility

The NCP suggests that a goal of the CERCLA process is to meet a site-wide cumulative acceptable risk level (EPA 1990a). However, EPA historically has been forced to address such considerations as technical feasibility, verification, uncertainty and cost in promulgating concentration limits for air (Clean Air Act) and water (Safe Drinking Water Act). In both cases, consideration for using best available technology (BAT) is written into the regulation. BAT is (40CFR141.2):

> "that technology, treatment or other means which the Administrator finds, after examination for efficacy under field conditions and not solely under laboratory conditions, are available (taking cost into consideration)" (EPA 1989h).

The NRC has relied on a similar concept, "As low as is reasonably achievable" (ALARA) in several promulgated regulations. ALARA allows for:

> "making every reasonable effort to maintain exposures to radiation as far below the dose limits ... taking into account the state of technology, the economics of improvement in relation to state of technology, the economics of improvement in relation to benefits to the public health and safety, and other societal and socioeconomic considerations" (10CFR20.3, NRC 1991).

Researchers have suggested that these concepts must begin to play a larger role in CERCLA cleanup efforts (Travis and Doty 1990). For example, groundwater scientists have predicted it may take as long as 100 to 200 years to lower contaminant concentrations in groundwater by a

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factor of 100 (Mackay and Cherry 1989; Hall 1988). EPA recently concluded that pumping and treating groundwater aquifers has resulted in significant mass removal, target levels (usually based on MCLs) have not been achieved at any CERCLA sites (EPA 1989j; Travis and Doty 1990). This suggests that technologies for remediating groundwater may not be capable of achieving ARAR-based RGs, much less the lower risk-based goals.

10.1.3.2 Verification

Two issues are important for discussing verification of risk-based remediation goals, especially for radionuclides. First, risk-based remediation goals for many radionuclides are a fraction of natural background in some media and would not be verifiable in the presence of background levels. The radiation doses corresponding to the risk range of 10^{-4} to 10^{-6} are 2.3 to 0.02 mrem per year, respectively, using the EPA risk coefficient of 6.2×10^{-7} mrem^{-1} (EPA 1989b) and a 70-year exposure. Neither of these radiation doses is discernible from natural background radiation doses, which exhibit significant variations, but are approximately 300 mrem per year (including radon exposure) (NCRP 1987). More simply, 300.02 mrem is not discernible from 300 mrem.

The second issue concerns the cost and time required to conduct analytical verification at the concentrations corresponding to a lifetime risk of 10^{-6}. For example, the concentration of U-238 in drinking water corresponding to a risk from lifetime exposure via the drinking water pathway is a fraction of the routine analytical detection limit in RI/FS groundwater sample analytical results. Nonroutine or enhanced radiochemical and sample analytical techniques are capable of achieving lower detection limits at the expense of additional laboratory time and cost. These enhanced techniques generally are not practical for routine large-scale sample analytical needs, as would be the case to verify remediation of contamination at the FEMP.

10.1.3.3 Uncertainties in Risk Estimates

Risk-based remediation goals embody considerable uncertainty that can be avoided by using ARAR values. Risk assessment is a process based on numerous assumptions, models, and parameters, each of which has associated uncertainties. For example, current risk factors assume that any level of exposure to a carcinogen may result in cancer (i.e., there is no dose threshold for cancer causation). In addition, it is assumed that the relationship between dose and risk is linear. Numerous data indicate that these assumptions overestimate actual risk. Data are constantly being gathered and interpreted to better understand the relationship between dose and risk. This ongoing process produces a variety of risk factors from which risks are estimated. This point is

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extremely important when proposing risk-based standards since a specific dose could be deemed to correspond to an acceptable risk depending on which risk factor is used to relate dose and risk.

Other uncertainties are associated with assumptions about the exposure assessment. Again, acceptability may be dependent on whether the risk assessor assumes a 30-year exposure (time at one residence; EPA 1989a) or a 70-year lifetime exposure (conventional); and whether the risk assessor assumes exposure under current or future hypothetical land-use scenarios. For example, depending on the assumptions used, a 25 mrem dose limit may or may not be considered acceptable by NCP standards. The differences in risk estimates are even greater when they are based on an exposure assessment assuming future hypothetical land-use conditions (e.g., on-site resident farming) rather than current site conditions (i.e., industrial site with controlled access).

While risk assessment is useful in areas where <u>relative</u> risk values are helpful (e.g., for comparing alternatives for the FS process), it may not be suitable to use for use in developing <u>absolute</u> concentration values. In the former situation, uncertainties are common to all alternatives and thus are generally not of great importance. In the latter situation, the absolute uncertainties are significant.

10.1.3.4 <u>Historical Precedent</u>

Historical precedent is an important consideration in the process of selecting final remediation levels, assuming technical and policy considerations were reviewed in earliest decisions. To date, Records of Decision (RODs) have been issued for fewer than 15 sites having radioactive materials as the contaminants of concern. All of the sites have radium-226 as the principal radioactive contaminant (EPA 1988f, EPA 1989k, EPA 1990c). This is significant since the remediation goals for sites having radium-226 contamination are not derived from an acceptable risk or risk range. Remediation goals at these sites are based on standards promulgated in <u>Environmental Protection Agency Standards for Protection Against Uranium Mill Tailings</u> (40CFR192.12) (EPA 1983), as well as the maximum contaminant levels for radium-226, radium-228, and gross alpha particle radioactivity in community water systems in <u>Environmental Protection Agency National Primary Drinking Water Regulations</u> (40CFR141.15) (EPA 1989h).

At the Maxey Flats low-level radiation CERCLA site, the EPA proposed 25 mrem/year to the whole body as a preliminary remediation goal, based on a relevant and appropriate requirement specified in 10CFR61.41 (Clay and Guimond 1990). Using the EPA risk coefficient of 6.2×10^{-7} $mrem^{-1}$ (EPA 1989b) and assuming a 70-year exposure, the lifetime risk associated with this exposure would be 1×10^{-3}, which is above the CERCLA goal.

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10.1.3.5 Acceptable Risk

The EPA has stated that in the case of radiation exposure, "when an ARAR for a specific chemical (or in this case, a group of chemicals) defines an acceptable level of exposure, compliance with the ARAR will generally be considered protective, even if it is outside of the risk range (unless there are extenuating circumstances such as exposure to multiple contaminants)" (Clay and Guimond 1990). Despite the parenthetical phrase, this statement suggests that definitions of acceptable risk other than 10^{-4} to 10^{-6} may be allowable in the CERCLA process. Promulgated radiological dose limits are set forth in regulations that have been subjected to a rulemaking process which is forced to use protectiveness of human health as a major criterion. As stated earlier, the definition of health protectiveness is different than that used in the CERCLA process.

10.1.3.6 Conclusion

However, CERCLA was designed to be implemented in conjunction with other environmental laws (i.e., ARARs). A major problem arises when CERCLA goals (e.g., cleanup levels based on the 10^{-4} to 10^{-6}) are in conflict with these other laws. Chemical-specific standards promulgated under these laws generally are designed to regulate health risks to an acceptable level, which in several cases is greater than 10^{-4}. In other words, the definition of "acceptable risk" or "acceptable exposure" is inherently different in different pieces of legislation. Thus, while both ARARs and CERCLA risk-based criteria generally are considered health protective, the risk levels on which they are based are different. Many ARARs are based on technological limitations (e.g., MCLs) and thus often represent the most protective level that is actually achievable.

10.2 FEASIBILITY STUDY RISK ASSESSMENT CHARACTERIZATION

Risk assessment for the FS is performed during the detailed analysis of alternatives. Risk assessment activities conducted for the detailed analysis of alternatives are an integral part of a hierarchy of nine criteria for evaluation of alternatives. The EPA specifies that the following nine evaluation criteria be used to evaluate all remedial alternatives at CERCLA sites (EPA 1988a):

- Threshold Criteria
  -Overall protection of human health and the environment
  -Compliance with ARARs

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- Primary Balancing Criteria
  - -Long-term effectiveness and permanence
  - -Reduction of toxicity, mobility, or volume
  - -Short-term effectiveness
  - -Implementability
  - -Cost

- Modifying Criteria
  - -State acceptance
  - -Community acceptance

The risk assessment for the detailed analysis of alternatives will provide input for three of the nine EPA evaluation criteria: overall protection of human health and the environment; long-term effectiveness and permanence; and short-term effectiveness of remedial alternatives.

As suggested in EPA guidance (EPA 1991h), the general methodology for evaluating long-term risks associated with remedial alternatives follows the methods used to determine baseline risks:

- Determine contaminants of concern identified in the baseline risk assessment which are associated with each alternative.

- Determine potential long-term and short-term exposure pathways and receptors associated with each alternative. EPA provides direction on some potentially significant contaminant transport mechanisms associated with common remedial alternatives (EPA 1991h).

- Estimate exposure and risks associated with each pathway, either quantitatively or qualitatively.

10.2.1 Overall Protection of Human Health and the Environment

Evaluation of the overall protection of human health and the environment for the remedial alternatives is based on long-term and short-term effectiveness of the remedial alternative in achieving the PRGs, and on compliance with ARARs. Overall protectiveness is a threshold criterion; alternatives that do not satisfy threshold criteria are eliminated from the alternative selection process (EPA 1988a).

10.2.2 Long-term Effectiveness and Permanence

The long-term effectiveness criterion addresses the ability of an alternative to protect human health and the environment from residual waste or hazardous materials that remain on site after completion of remediation. From a risk perspective, this criterion is concerned with quantifying the magnitude of residual risks associated with remedial alternatives. Magnitude of residual risks will be quantitatively evaluated in the detailed analysis of alternatives by examination of potential exposures to individuals after remediation.

The FS risk assessment will quantify residual hazardous materials remaining after remediation, identify potential reasonable maximum exposed individuals, identify potential significant exposure pathways, and evaluate the risks to the RME individual as per EPA guidance (EPA 1991h). The long-term effectiveness criterion will be evaluated for all the alternatives with two exposure scenarios: one assuming DOE will retain control of the property, the other assuming use of the site by a resident farmer. For the no-action alternative, risks will be assessed with and without institutional controls.

10.2.3 Short-term Effectiveness

The short-term effectiveness criterion addresses the risk from exposure to waste or hazardous materials as a result of implementing a remedial alternative. From a risk perspective, this criterion is concerned with quantifying the potential magnitude of exposure and risk to the community, to workers and the environment during remediation.

Where potential exposure pathways that are unique to implementation of a remedial alternative are identified, an assessment methodology will be devised to perform either a qualitative or quantitative assessment for the alternative. Specific methods used to estimate a remedial alternative risks are discussed in Sections 5.0 through 7.0 for each identified pathway.

10.2.3.1 Risks to the Public During Remediation

Evaluation of the degree of risk to the public during remediation involves similar potential receptors and exposure pathways as under baseline conditions. However, acute or sub-chronic exposures are of greater concern during remediation than chronic exposures. Also, exposure concentrations and exposure durations during remediation differ from those under baseline conditions. Pathways to be evaluated include, but are not limited to, the following:

- Risk to the public from transportation accident injuries and fatalities during transportation of waste to an off-site disposal facility
- Airborne releases due to disturbance of contaminants that pose a potential inhalation hazard
- Increased surface water runoff from disturbing compacted soils and ground cover.

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For evaluation of exposures to the public under short-term effectiveness, it is assumed that existing security controls and institutional controls at the property boundary restrict access to the site. This assumption is made for all alternatives, other than the no-action alternative, with respect to the short-term effectiveness evaluation.

10.2.3.2 Risks to Workers During Remediation

Evaluation of the risks to workers during remediation is considered separately from evaluation of risk to the community. The separation is appropriate because of the need to assess transient exposures to workers who are closer to the hazardous wastes and the remediation activities than are members of the community. This proximity to the site potentially subjects the workers to more acute exposure situations. Because of the potential for more acute exposures, worker protection and engineering considerations incorporated into remedial alternatives will include consideration of the "As Low As Reasonably Achievable" (ALARA) principle to optimize exposure and risk. Assessment of risks to remediation workers will be performed for the following pathways:

- Exposure to penetrating gamma radiation fields
- Exposure to contaminants via dermal contact during nonroutine events
- Exposure to airborne contaminants via inhalation
- Risk of transportation accident injury and fatality
- Risk of construction accident injury and fatality

The degree of protection of on-property workers during remediation will be evaluated with respect to occupational limits rather than the acceptable range of lifetime health risk in the NCP (EPA 1990a). Occupational exposure standards are implemented in the site Health and Safety Program and control exposure to hazardous materials for on-property workers. Worker exposures to contaminants during remediation will be calculated using methods described in preceding sections. Methods for calculating risk from construction and transportation activities are described below.

Construction Risks

General risks associated with construction operations will be estimated for each alternative using historical risk data. The construction work risks are calculated in the following manner:

$$\text{Risk} = (PH)(RC) \qquad (10\text{-}13)$$

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where

Risk = Risk of injury or fatality expressed as a probability
PH = Person-hours of construction work
RC = Injury or fatality risk coefficient (risk/person-hour)

Risk factors used are from the U.S. Department of Labor (1988):

- 3.4×10^{-5} injuries per man-hour
- 5.0×10^{-7} fatalities per man-hour

<u>Transportation Risks</u>

Since remedial actions calling for off-site disposal involve stabilization of the packaged waste, no exposures to hazardous materials are expected to occur during transportation. However, the potential exists for highway deaths and accidents to occur. For each alternative involving off-site disposal, the following method will be used to calculate transportation risks:

- Estimates will be made of the total volume of waste to be transported off site.
- Using density estimates, the total weight (in pounds) will be estimated.
- The estimated weight will be used to determine the number of shipping containers required to ship the wastes.
- Values for containers per truckload will be used to determine the number of truckloads or rail loads required to transport the total volume of waste.

$$\text{Risk} = (N)(CF)(RC) \qquad (10\text{-}14)$$

where

Risk = Risk of injury or fatality expressed as a unitless probability
N = Number of round trips made
CF = Mileage per round trip
RC = Injury or fatality risk coefficient (risk/mile)

Department of Transportation (DOT) and Nuclear Regulatory Commission (NRC) regulations were reviewed to determine proper shipping containers and loads (DOT 1989; NRC 1989). Table 10-4 lists the specific parameters that will be used to calculate transportation risks.

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TABLE 10-4
PARAMETERS USED TO CALCULATE TRANSPORTATION RISKS

Parameters	Value	Reference/Justification
<u>Waste Mass</u>		
To be determined specifically for each operable unit and remedial action alternative		
<u>Shipping Capacities</u>		
LSA container box	90 ft^3, or 9000 lbs	Manufacturer specifications
Maximum/truck	40,000 lbs	
Gondola capacity	70 tons/car	
Train capacity	10 cars/trip 90 cars/trip	Assuming non-exclusive use of the train. Assuming exclusive use of the train.
<u>Round trip mileage to Disposal Site</u>		
Truck	4400 miles	Three sites were considered as potential disposal sites: the Hanford site, Richland, WA, the Nevada Test Site (NTS), NV, and Envirocare, Clive, UT. Mileage was determined for each site. Mileage to NTS was used for calculations since it was the mid-range of the three sites.
Rail	4550 miles	Same as above.
<u>Risk Factors - Truck Transport</u>	<u>Fatalities/ Mile</u>	
Occupational Driver Fatalities	2.1E-9	DOT 1986; FHA 1988; Statistics are for "authorized carrier" which is an interstate carrier
Occupational Driver Injuries	4.1E-8	Same as above.

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TABLE 10-4
(Continued)

Public Fatalities	1.3E-8	DOT 1986; FHA 1988; "Public" includes passengers in trucks, driver and passengers in cars, pedestrians, etc.
Public Injuries	1.2E-7	Same as above.
<u>Rail Transport</u>		
Employee Fatalities	4.6E-8	DOT 1988
Employee Injuries	4.6E-6	DOT 1988
Public Fatalities	1.8E-6	DOT 1988; "Public includes train passengers, off-duty workers, pedestrians, drivers and passengers in other vehicles, etc.
Public Injuries	6.8E-6	Same as above.

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10.2.3.3 Toxicity Assessment and Risk Characterization

Because of the short duration of exposure during remediation, subchronic RfDs will be used to evaluate noncarcinogenic effects. If available, toxicity information based on short-term exposures will be used for carcinogens. Such toxicity information may include acute inhalation criteria (AIC), minimal risk levels (MRLs), threshold limit values (TLVs), and permissible exposure levels (PELs).

The risk characterization for carcinogens will involve comparing calculated intakes to short-term toxicity values. Radionuclide risks will be calculated using slope factors. In addition, doses will be calculated in order to compare exposures to short-term dose limits.

10.2.4 Risk Assessment for an On-Site Waste Management Facility

Construction and operation of an on-site waste management facility is an integral part of numerous remedial alternatives under consideration for the FEMP. Therefore, risk assessment concerns potentially associated with such a facility must be addressed in the site-wide FS risk assessment. The area under consideration for an on-site waste management facility lies north and east of the production area within the FEMP property boundary.

Risks potentially associated with the on-site waste management facility are divided into three categories:

- The baseline risk scenario (before construction)
- The short-term risk scenario (during construction and placement of waste)
- The long-term risk scenario (during storage of waste)

The methodology for assessing risks potentially associated with the on-site waste management facility is consistent with the methodology described in preceding sections of this Addendum.

10.3 SITE-WIDE OPTIMIZATION MODEL

As a part of addressing site-wide risk concerns, an optimization model will be used to optimally track allowable residual risks among operable units. The model will be a tool that will help risk managers select the optimal remediation alternative for each operable unit, as each operable unit moves through a staggered FS process (see Section 2.0). The model will:

- Use preliminary risk estimates in the early stages of the process
- Add final risk estimates as they become available
- Use ARARs as well as risk constraints

The risk assessment/risk management model will:

- Minimize site-wide cleanup cost while constraining site-wide risk so that the sum of the risks from each operable unit does not exceed a predetermined acceptable site-wide risk level.

- Track the matrix of alternatives for all operable units as preliminary information is available about engineering alternatives and associated risks to insure that all residual contamination remaining after treatment meets an acceptable site-wide risk goal.

- Make information available on multiple alternative selection scenarios across operable units to give risk managers several options for meeting the site-wide residual risk goal. This will allow risk assessors to recommend the best alternative for a given operable unit from a site-wide risk perspective and minimize the chance that an alternative selected during the first operable unit FS process will have to be changed once all operable unit FS processes are complete.

- Supply risk assessors and risk managers with:

  - Information on site-wide risk consequences associated with selecting an alternative for a single operable unit (e.g., the limitations that a selection places on other operable units)

  - Information to help select the best alternative for operable units yet to proceed through the FS

  - Information on the uncertainties associated with risk assessment data and a description of how these uncertainties could affect the selection of a particular alternative

Six steps are involved in implementing the site-wide optimization approach:

1) Develop the preliminary model.

2) Estimate preliminary risk and cost associated with each alternative for each operable unit and input results in the model.

3) Run the model using preliminary risk and cost estimates.

4) Determine the risk associated with the selected alternative for the first operable unit to proceed through the FS process. Update the model's input data, and run the model again. Repeat this task after each subsequent operable unit FS.

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5) Provide output to risk managers as the FS processes progresses, ensuring that an operable unit alternative selection does not adversely constrain the options available for subsequent operable units.

The model will be used to track site-wide risk concerns as each operable unit moves through the FS process depicted in Figure 2-2. Note that as RODs are written for the initial operable units, the selected alternative will be the only alternative that remains as part of the model.

The major assumptions that will be used while performing the optimization task are:

- All operable units pose a risk to human health and the environment.
- The risks from all operable units are additive.

It is conservative to assume that total site risk is the sum of all operable unit risks, since many pathways to the site-wide reasonable maximum exposure are for various operable units, and thus would not be additive. However, this assumption of additivity should prevent the sum of the individual operable unit risks from exceeding the site-wide residual risk limit. In addition, summing the small risk values (e.g., 1×10^{-6} and 1×10^{-7}) associated with most alternatives other than the no action alternative most likely will not effect the outcome of the modeling.

Tables 10-5 and 10-6 provide <u>example</u> model input for the preliminary model currently under consideration the example is for Operable Unit 1 remedial alternatives. The model software is a linear programming model called LINDO (Schrage 1991) that is routinely applied for operational research and industrial cost optimization. It allows input of one objective parameter and up to 100 constraints and 200 variables on which to perform an optimization of the objective. In the example problem, cost minimization is the objective and risk is the constraint. The sum of the risks of a single operable unit can not exceed (1×10^{-6}). Additional criteria considered in the model include the balancing criteria required for remedial action decision-making. Ranking values from 1 to 10 are used to describe these semi-qualitative parameters.

Data output from the model includes the optimal solution (e.g., the best solution) plus several types of sensitivity analyses (not included in this data file). This sensitivity information includes the range that the risk constraint (10^{-6}) may vary before the optimal solution would change, the amount that cost for each alternative may vary before the optimal solution would change. This type of sensitivity information is important when dealing with preliminary data. The preliminary model is being used to address the requirements of the Amended Consent Agreement, which

TABLE 10-5
EXAMPLE MODEL INPUT FOR OPERABLE UNIT 1

	Alternative 0	Alternative 2	Alternative 4A	Alternative 4B	Alternative 5A	Alternative 5B	Alternative 6	Alternative 7
Description	No Action Alternative	Nonremoval, stabilization, slurry wall, cap	Removal, treatment (cement), on-site disposal	Removal, treatment (vitrificaiton), on-site disposal	Removal, treatment (cement), on-site disposal	Removal, treatment (vitrification), on-site disposal	Removal, treatment, on-site disposal, no soil treatment, cap	Removal, treatment, in-situ soil treatment, on-site disposal, cap
Cost in Dollars	10.0	9.6	5.7	2.2	1.0	5.6	7.1	5.4
Long-Term Risk	1×10^0	2.6×10^5	7×10^7 assumed	7×10^7 assumed	5×10^{-7} assumed	5×10^7 assumed	7×10^7	2.0×10^6
Reduction of Toxicity, Mobility, and Volume through Treatment	1	3	8	9	10	10	7.5 - Vitrification 6 - Cement	8.5 - Vitrification 7 - Cement
Ranking of Short-Term Effectiveness	10	8	7	5	6	3	2 - Vitrification 4 - Cement	1 - Vitrification 3 - Cement
Ranking of Long-Term Effectiveness: Reliability of Controls	1	3	7	8	10	10	6.5 - Vitrification 5.5 - Cement	7.5 - Vitrification 6.5 - Cement
Implementability	10	9	8	4	7	3	5 - Vitrification 7.5 - Cement	1 - Vitrification 6.5 - Cement

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TABLE 10-6
EXAMPLE MODEL INPUT VERIFICATION FOR OPERABLE UNIT 1

	Alternative 0	Alternative 2	Alternative 4A	Alternative 4B	Alternative 5A	Alternative 5B	Alternative 6	Alternative 7
Description	No Action Alternative	Nonremoval, stabilization, slurry wall, cap	Removal, treatment (cement), on-site disposal	Removal, treatment (vitrificaiton), on-site disposal	Removal, treatment (cement), on-site disposal	Removal, treatment (vitrification), on-site disposal	Removal, treatment, on-site disposal, no soil treatment, cap	Removal, treatment, in-situ soil treatment, on-site disposal, cap
Cost in Dollars	9,000,000	158,220,000	1,565,860,000	2,833,510,000	3,281,270,000	1,597,670,000	1,060,960,000	1,698,370,000
Long-Term Risk	1×10^{0}	2.6×10^{5}	7×10^{7} assumed	7×10^{7} assumed	5×10^{7} assumed	5×10^{7} assumed	7×10^{7}	2.0×10^{6}
Reduction of Toxicity, Mobility, and Volume through Treatment	No treatment, therefore, no reduction of toxicity, mobility or volume.	Treatment of the waste in this alternative consists of compaction which reduces the mobility of the contamination and the volume. There is no effect on the toxicity of the waste from this alternative.	Stabilization, supplemented by disposal in an engineered disposal facility (EDF), will provide a high level of reduction in mobility. The volume will be increased by approximately 30% to 1.6 million yd^3. Toxicity is reduced appreciably by stabilization.	The waste will be vitrified thus reducing to a high degree the waste mobility and toxicity. The volume to be treated is approximately 1.6 million yd^3 and there will be a volume reduction of about 30%.	Stabilization of the waste by cement provides a high degree of reduction in mobility and toxicity. No residual waste will remain on site. The volume of the waste will increase by about 30%.	Stabilization of the waste by vitrification provides a high degree of reduction in mobility and toxicity. No residual waste will remain on site. The volume of the waste will decrease by about 30%.	The waste will be stabilized by either cement or vitrification and will reduce the mobility and toxicity to a great degree. The vitrification process will reduce the mobility and toxicity to a greater degree than cement. Use of the vitrification process will result in a decrease of about 30% in volume while use of cement will result in an increase of about 30%.	The waste material is to be stabilized and/or vitrified which will greatly reduce the mobility and toxicity of the waste. Cement stabilization will increase the volume of the waste by as much as 50% and vitrification will reduce the waste volume by about 30%.

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TABLE 10-6
(Continued)

Description	Alternative 0	Alternative 2	Alternative 4A	Alternative 4B	Alternative 5A	Alternative 5B	Alternative 6	Alternative 7
	No Action Alternative	Nonremoval, stabilization, slurry wall, cap	Removal, treatment (cement), on-site disposal	Removal, treatment (vitrificaiton), on-site disposal	Removal, treatment (cement), on-site disposal	Removal, treatment (vitrification), on-site disposal	Removal, treatment, on-site disposal, no soil treatment, cap	Removal, treatment, in-situ soil treatment, on-site disposal, cap
Long-Term Effectiveness: Reliability of Controls	This alternative does not provide a long term solution that is effective and permanent. The magnitude of the risk is not reduced from the current level.	This alternative will virtually eliminate direct radiation exposure from the pits. Potential exposure of the pit wastes via groundwater is a concern. This alternative requires maintenance in perpetuity. Monitoring wells will be required to be replaced periodically. Since waste is left on site, review of the remedy will be required every five years per CERCLA Section 121(c).	Risk from on-property waste disposal is reduced from baseline risk. The EDF requires long term maintenance and use of groundwater monitoring wells which will require periodic replace-ment. The leachate collection tank will require monitoring and removal of collected leachate. Since waste is left on site, review of the remedy will be required every five years per CERCLA Section 121(c).	Some residual risk is associated with disposal of the waste in the EDF. The EDF requires long term maintenance and use of groundwater monitoring wells which will require periodic replace-ment. The leachate collection tank will require monitoring and removal of collected leachate. Since waste is left on site, review of the remedy will be required every five years per CERCLA Section 121(c).	The residual risk associated with the waste material is eliminated because the waste is shipped off-site.	The residual risk associated with the waste material is eliminated because the waste is shipped off-site.	Some residual risk from on-site disposal exists, but is greatly reduced from the baseline residual risk. A minor potential for exposure from contaminated soil via groundwater is present. Due to the small amounts of radioactive contaminants present in the soils and the existence of the cap, direct radiation exposures from the pits will be virtually eliminated. The cap will cover and contain any contaminated soil around the pits, but will require mainten-ance in perpetuity. Monitoring wells will be required and will have to be replaced periodically. Since waste is left on site, review of the remedy will be required every five years per CERCLA Section 121(c).	The treatment of the soil by in-situ stabilization will limit the spread of radioactive material via leaching, volatizing, etc. Radioactive and hazardous components will be contained within a solid matrix with the radioactive constituents contained long enough to decay to its daughters. The installation of the cap will further serve to impede the spread of contamination from the area of concern. The remainder of the long term effectiveness is the same as Alternative 6.

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TABLE 10-6
(Continued)

	Alternative 0	Alternative 2	Alternative 4A	Alternative 4B	Alternative 5A	Alternative 5B	Alternative 6	Alternative 7
Description	No Action Alternative	Nonremoval, stabilization, slurry wall, cap	Removal, treatment (cement), on-site disposal	Removal, treatment (vitrificaiton), on-site disposal	Removal, treatment (cement), on-site disposal	Removal, treatment (vitrification), on-site disposal	Removal, treatment, on-site disposal, no soil treatment, cap	Removal, treatment, in-situ soil treatment, on-site disposal, cap
Short-Term Effectiveness	The environmental effects attributable to the waste pits would continue.	There will be minimal impact on short term effectiveness from this alternative because the pit waste is treated in- situ. There is a slight possibility of fugitive dust, fumes and odors from construction of the cap. Paddys Run will have to be rerouted causing short term surface water runoff turbidity. Impacts on Paddys Run biota will be short term loss of riparian and aquatic and associated species. There will be similar impacts to terrestrial species due to construction of the cap and slurry wall. There also will be increased traffic, noise and road degradation from the cap and slurry wall construction.	There is increased probability of an accidental release of uranium, thorium, and radon due to waste removal, but is minimized by conducting the operations in a controlled environment. Minor amounts of fugitive dust, fumes and odors are associated with heavy equipment operations during construction of the waste treatment, packaging and EDF and from transportation of the waste to the on-site EDF. There are possible impacts on aquatic and terrestrial species associated with siting of the EDF. Transportation of construction materials for the EDF causes increased traffic congestion, noise and road degradation. Protection and minimization of exposure to workers will be ensured by the waste processing building and health & safety measures. Construction fatalities and injuries are possible due to the high labor hours for the construction of the EDF.	There is increased probability of an accidental release of uranium, thorium, and radon due to waste removal, but is minimized by conducting the operations in a controlled environment. Minor amounts of fugitive dust, fumes and odors are associated with heavy equipment operations during construction of the waste treatment, packaging and EDF and from transportation of the waste to the on-site EDF. There are possible impacts on aquatic and terrestrial species associated with siting of the EDF. Transportation of construction materials for the EDF causes increased traffic congestion, noise and road degradation. Protection and minimization of exposure to workers will be ensured by the waste processing building and health & safety measures. Construction fatalities and injuries are possible due to the high labor hours for the construction of the EDF.	There is increased probability of an accidental release of uranium, thorium, and radon due to waste removal, but is minimized by conducting the operations in the EIE and the waste processing building. Minor amounts of fugitive dust, fumes and odors are associated with heavy equipment operations during construction of the waste treatment, packaging buildings and from rail transportation of the waste to the off-site EDF. There are possible impacts on aquatic and terrestrial species associated with siting of the support facilities. Transportation of hazardous and radioactive waste to an off-site disposal facility could cause rail congestion and increased noise. Risk to workers and the community is associated with off-site transportation of waste.	Short-term effects will be the same as Alternative 5A except for the additional possibility of gaseous, toxic and radioactive emissions associated with the vitrification process. The scrubbers used for the treatment of the off gases would require a water supply which will impact surface and groundwater varying with the amount of water required.	Short-term effects for this alternative are the same as for Alternative 4 with the additional treatment of standing water will generate contaminated sludges/resins that will have to be disposed of in accordance with standard waste management practices. There are possible impacts on groundwater, including the sole-source aquifer, associated with any failures of leachate collection systems in the waste disposal system.	The short-term effects of this alternative is the same as Alternative 6 with the additional risk of migration of contaminants via fugitive dust from the in-situ soil mixing.

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TABLE 10-6
(Continued)

Description	Alternative 0 No Action Alternative	Alternative 2 Nonremoval, stabilization, slurry wall, cap	Alternative 4A Removal, treatment (cement), on-site disposal	Alternative 4B Removal, treatment (vitrificaiton), on-site disposal	Alternative 5A Removal, treatment (cement), on-site disposal	Alternative 5B Removal, treatment (vitrification), on-site disposal	Alternative 6 Removal, treatment, on-site disposal, no soil treatment, cap	Alternative 7 Removal, treatment, in-situ soil treatment, on-site disposal, cap
Implementability	This action is easy to implement because there is nothing to implement with the exception of an improved monitoring system.	Caps and slurry walls are routinely constructed and have been used at other DOE sites. Surcharging pits to decrease void space has been used on stabilization projects in the past. No delays are expected due to technical problems.	Stabilization technology is proven, but requires treatability tests. Future remedial action will likely add new protective layers to the EDF. Monitoring consists of visual inspection, leachate detection and groundwater sampling.	Batch and continuous vitrification technology has been proven on a small scale (about 100 yd^3/d) There are no units available to meet the OU1 remediation goals in a timely manner (6 to 20 yrs) Future remedial action will likely add new protective layers to the EDF. Monitoring consists of visual inspection, leachate detection and groundwater sampling.	Cement solidification is proven technology, but treatability tests are required. Because the waste is shipped off-site, future remediation and monitoring is not required. Transportation to the off-site depository most likely will require state, local and federal agency approvals and extensive coordination. The technology, equipment and associated technical specialists are readily available for the remedial action.	Batch and continuous vitrification technology has been proven on a small scale (about 100 yd^3/d) There are no units available to meet the OU1 remediation goals in a timely manner (6 to 20 yrs) Because the waste is shipped off-site, future remediation and monitoring is not required. Transportation to the off-site depository most likely will require state, local and federal agency approvals and extensive coordination.	Batch and continuous vitrification technology has been proven on a small scale (about 100 yd^3/d) Cement stabilization batch plants have been used on large scale bases and are slightly more implementable, but require treatability studies. Both technologies if used together would result in a high level of implementability.	Batch and continuous vitrification technology has been proven on a small scale (about 100 yd^3/d) Cement stabilization batch plants have been used on large scale bases and are slightly more implementable, but require treatability studies. Future remedial action will likely add new protective layers to the EDF. Monitoring consists of visual inspection, leachate detection and groundwater sampling.

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states that preliminary leading remedial alternatives will be selected for each operable unit. In the early stages of model development (e.g., prior to complete site scoping activities and prior to generating data on each alternative), model output will be of limited use. However, the model will be useful in helping to direct all FS activities from a site-wide risk perspective. As more data are obtained, the model will become finalized and will be useful for performing cost-risk optimization.

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Attachment 1

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ATTACHMENT I

GENERAL OUTLINE
FOR A BASELINE RISK ASSESSMENT REPORT
FOR THE RI/FS AT THE FEMP

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1.0 INTRODUCTION

Include a brief discussion of why the RI/FS is being performed at the FEMP.

1.1 Risk Assessment Objectives

- Definition of the objectives of the specific RI/FS baseline risk assessment of interest.

1.2 Organization of Risk Assessment Report

- Brief description of the organization of the specific RI/FS baseline risk assessment of interest, including general content of major sections.

1.3 Site Background

- Brief reference to the appropriate remedial investigation report or the site-wide characterization report for information pertaining to site physical description, general site history, general descriptions of local populations, and general descriptions of sampling efforts.

- Brief reference to the risk assessment work plan addendum for discussion of the approach to completion of risk assessments for the RI/FS under new Consent Agreement modifications.

2.0 IDENTIFICATION OF CONSTITUENTS OF POTENTIAL CONCERN

2.1 General Site-Specific Data Collection and Evaluation Considerations

- Brief reference to the appropriate remedial investigation report or the sitewide characterization report for information pertaining to data collection and evaluation activities.

- Brief reference to the risk assessment work plan addendum for discussion of site-specific methods for evaluation of analytical results, determination of background levels of constituents, and determination of constituents of potential concern for risk assessment.

2.2 Selection of Constituents of Potential Concern

- Reiterate selection criteria for determining constituents of potential concern

- Presentation of actual constituents of potential concern for quantitative evaluation in the risk assessment.

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3.0 EXPOSURE ASSESSMENT

3.1 Characterization of Exposure Setting

Include a brief summary of similar material in remedial investigation report or site-wide characterization report.

- Physical Setting
  - Climate
  - Vegetation
  - Soil type
  - Surface hydrology
  - Groundwater hydrology

- Potentially Exposed Populations
  - Relative locations of populations with respect to site
  - Current land use
  - Potential alternate future land uses
  - Subpopulations of potential concern

3.2 Identification of Exposure Pathways

- Sources and receiving media
- Fate and transport in release media
- Exposure points and exposure routes
- Integration of sources, releases, fate and transport mechanisms, exposure points, and exposure routes into complete exposure pathways
- Summary of exposure pathways to be quantified in this assessment

3.3 Quantification of Exposure

- Exposure concentrations
- Estimation of constituent intakes for individual pathways

3.4 Identification of Uncertainties

- Current and future land-use
- Environmental sampling and analysis
- Exposure pathways evaluated
- Fate and transport modeling
- Parameter values

3.5 Summary of Exposure Assessment

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4.0 TOXICITY ASSESSMENT

4.1 Toxicity Information for Noncarcinogenic Effects

- Appropriate exposure periods for toxicity values
- Up-to-date RfDs for all chemicals
- One- and ten-day health advisories for shorter-term oral exposures
- Overall data base and the critical study on which the toxicity value is based (including the critical effect and the uncertainty and modifying factors used in the calculation)
- Effects that may appear at doses higher than those required to elicit the critical effect
- Absorption efficiency considered

4.2 Toxicity Information for Carcinogenic Effects

- Exposure averaged over a lifetime
- Up-to-date slope factors for all carcinogens
- Weight-of-evidence classification for all carcinogens
- Type of cancer for Class A carcinogens
- Concentration above which the dose-response curve is no longer linear

4.3 Chemicals for which No EPA Toxicity Values are Available

- Review of ECAO
- Qualitative evaluation
- Documentation/justification of any new toxicity values developed

4.4 Uncertainties Related to Toxicity Information

- Quality of the individual studies
- Completeness of the overall data base

4.5 Summary of Toxicity Information

5.0 RISK CHARACTERIZATION

5.1 Current Land-Use Conditions

- Carcinogenic risk of individual substances
- Chronic hazard quotient calculation (individual substances)
- Subchronic hazard quotient calculation (individual substances)
- Short-term hazard quotient calculation (individual substances)
- Carcinogenic risk (multiple substances)
- Chronic hazard index (multiple substances)
- Subchronic hazard index (multiple substances)
- Short-term hazard index calculation (multiple substances)
- Segregation of hazard indices

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- Justification for combining risks across pathways
- Noncarcinogenic hazard index (multiple pathways)
- Carcinogenic risk (multiple pathways)

5.2 Future Land-Use Conditions

- Carcinogenic risk of individual substances
- Chronic hazard quotient calculation (individual substances)
- Subchronic hazard quotient calculation (individual substances)
- Carcinogenic risk (multiple substances)
- Chronic hazard index (multiple substances)
- Subchronic hazard index (multiple substances)
- Segregation of hazard indices
- Justification for combining risks across pathways
- Noncarcinogenic hazard index (multiple pathways)
- Carcinogenic risk (multiple pathways)

5.3 Uncertainties

- Site-specific uncertainty factors
  - Definition of physical setting
  - Model applicability and assumptions
  - Parameter values for fate/transport and exposure calculations
- Summary of toxicity assessment uncertainty
  - Identification of potential health effects
  - Derivation of toxicity value
  - Potential for synergistic or antagonistic interactions
  - Uncertainty in evaluating less-than-lifetime exposures

5.4 Summary Discussion and Tabulation of the Risk Characterization

- Key site-related contaminants and key exposure pathways identified
- Types of health risk of concern
- Level of confidence in the quantitative information used to estimate risk
- Presentation of qualitative information on toxicity
- Confidence in the key exposure estimates for the key exposure pathways
- Magnitude of the carcinogenic and noncarcinogenic risk estimates
- Major factors driving risk
- Major factors contributing to uncertainty

6.0 ECOLOGICAL ASSESSMENT

6.1 Objectives of the Ecological Assessment
6.2 Scope of the Ecological Assessment
6.3 Ecological Description of Study Area
6.4 Constituents of Concern
6.5 Characterization of Exposure
6.6 Characterization of Risk
6.7 Quantitative Risk Characterization

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7.0 SUMMARY

7.1 Chemicals of Potential Concern
7.2 Exposure Assessment
7.3 Toxicity Assessment
7.4 Risk Characterization
7.5 Ecological Assessment